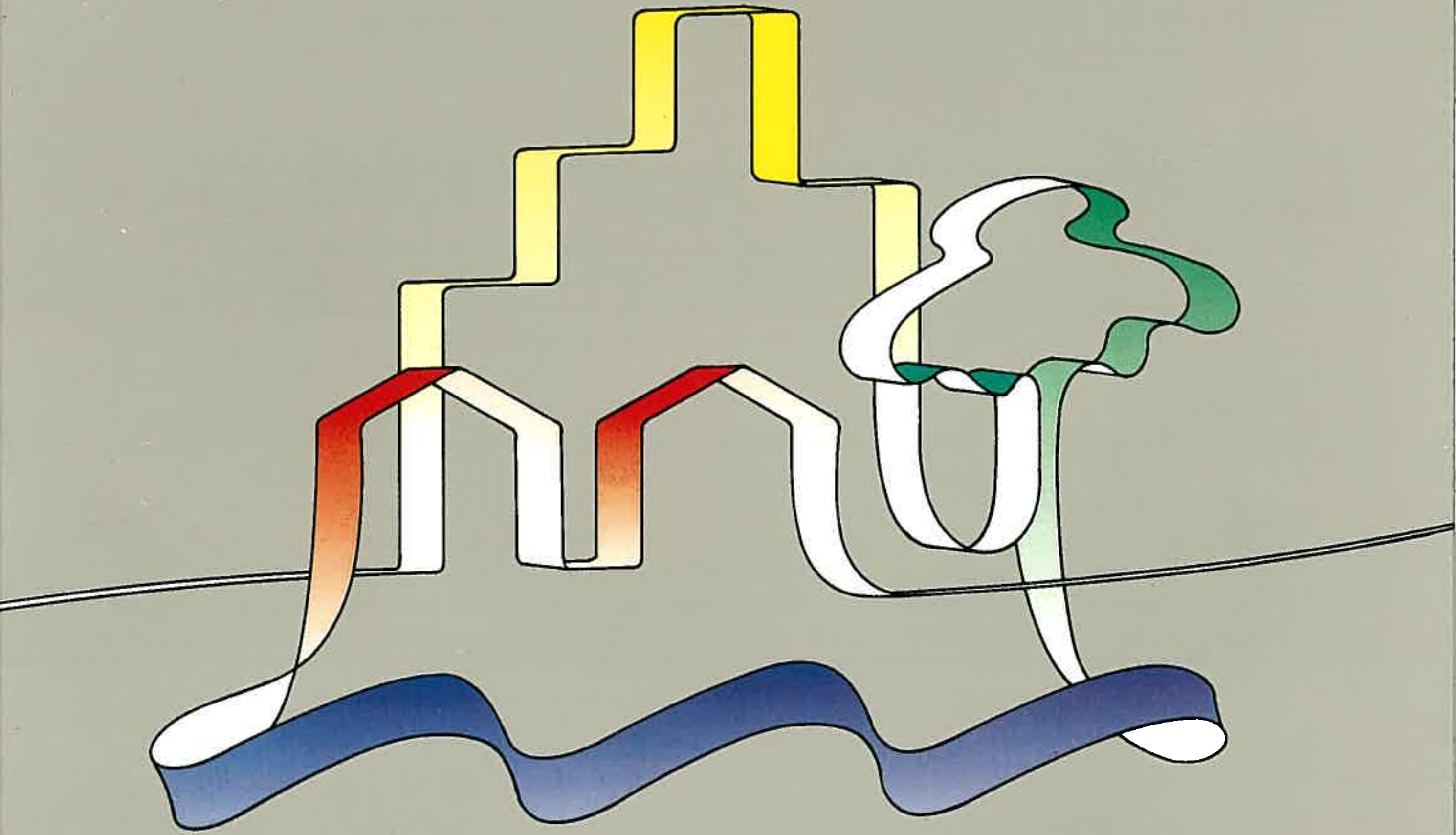
Urban Water Research Association of Australia

Impact of Urban Lawns on Nutrient Contamination of an Unconfined Aquifer



Research Report No. 49

URBAN WATER RESEARCH ASSOCIATION OF AUSTRALIA

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Impact of Urban Lawns on Nutrient Contamination of an Unconfined Aquifer

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FOREWORD

This report is based on UWRAA Research Project No WR-15: 'Impact of urban land uses on nutrient contamination of unconfined aquifers' which was undertaken during the period July 1991 - December 1992. Organisational responsibility for the project was as follows:

Sponsoring Authority : Water Authority of Western Australia

Research Agency : CSIRO Division of Water Resources

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The project was funded by the Urban Water Research Association of Australia and the Water Authority of Western Australia.

SYNOPSIS

Experimental and modelling approaches were developed and used to quantify water and nutrient leaching beneath urban lawns, situated on a Coastal Plain.

Four representative lawns were studied. Over a 210 day period, irrigation was the main source of water during summer, precipitation was the main source during winter. There was a strong seasonality in nitrogen and phosphorus concentrations in water leaching through the soil. Fertilising is conducted almost exclusively over spring and summer. About 50% of input water passed below the root zone, carrying nutrients with it. On many occasions, nitrate concentrations in leachate exceeded the World Health Organisation (WHO) drinking water limit of 10 mg/l. Invariably groundwater concentrations, however, were lower due to dilution and denitrification.

Substantial quantities of nitrogen and phosphorous have been shown to leach below the root zone and are a threat to many urban wetlands, especially the Swan-Canning river system.

Mechanistic modelling approaches were employed to simulate chloride and nutrient leaching below the root zone. A reasonable comparison between measured and modelled NH_4 and NO_3 values was observed. Measurements and modelling showed that although some PO_4-P passed below the root zone, it will take many years for the phosphate peak to reach the groundwater.

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1. BACKGROUND AND INTRODUCTION

Urbanisation has long been recognised as a cause of contamination of surface and ground water resources. Nutrient sources in urban areas include lawn and garden fertilisers, septic tanks, leaky sewer lines, urban runoff and waste disposal sites (Kimmel 1984).

Eutrophication of water bodies and waterways due to high nutrient levels (especially phosphorus and nitrogen) can cause undesirable algal blooms (Logan 1990). The threshold concentration of P for eutrophication lies between 0.02 to 0.05 mg/l. (Vollemweiller 1983; Ryding and Rast 1989). From a human perspective, high NO_3 is of major concern. The suggested World Health Organisation (WHO) limit for NO_3-N in drinking water is 10 mg/l.

The Perth Metropolitan Area is situated on a sandy coastal plain and contains extensive areas of lawn in domestic and public open spaces. The unconfined aquifers of the Swan Coastal Plain (WA) are a major source of the public water supply, meeting up to 40% of the Perth metropolitan domestic demand. In addition private and irrigation use far exceeds the public supply (Webster 1989). Because of the low water and nutrient retention capacity of coastal sands, the potential for nutrient leaching is high. Often horticultural and urban developments are closely integrated and these two land uses are major causes of nutrient contamination of groundwaters (Gerritse *et al.*, 1990; Sharma *et al.*, 1991). The consequences on nutrient leaching to aquifers of high fertiliser and manure inputs and heavy irrigation in the horticultural industry, have been addressed in earlier studies (Pionke *et al.*, 1990; Sharma *et al.*, 1991, 1992).

Septic tanks and urban agriculture (lawns, gardens) account for the greatest documented non-point source impact on the nitrogen status of the aquifer (Appleyard and Bawden 1987; Attwood and Barber 1989; Gerritse *et al.*, 1990). The NO_3, PO_4 concentrations leaching below the root zone under intensive horticulture can be comparable with those measured in the leachates of septic tanks (Whelan and Titamanis 1982; Whelan 1987; Sharma *et al.*, 1991) and which are similar to sewerage effluent ($N \simeq 40$ mg/l, $P \simeq 15$ mg/l). It is estimated that the input of N and P through septic systems in a housing density of 10 houses/ha, can be of a similar order of magnitude as from fertiliser used for lawns and gardens (Whelan and Titamanis 1982; Gerritse *et al.*, 1990).

With increasing urbanisation in the Perth metropolitan area, there is a trend showing that the quality of groundwater pumped from some public bore fields has been adversely affected in terms of TDS and some NO_3. Most TDS consist of salts of chloride and sulphate (Appleyard and Bawden 1986; Attwood and Barber 1989; Gerritse *et al.*, 1990).

Urbanisation on the Coastal Plain is inevitable as accelerated expansion is planned (State Planning Commission 1987). With a projected population of 1.3 million by the year 2000 and a current residential density of 6-9 dwellings/ha, the Perth metropolitan area would cover an estimated 700 km^2. Since all future urban development will be centrally sewered, the primary source of nutrient contamination is likely to be fertilisers in urban agriculture.

It is estimated that some 5 000 tonnes/yr of fertiliser (Whelan 1987) is used in the Perth metropolitan area, which will provide about 400 tonnes/yr (8×10^5 kg) of nitrogen and 200 tonnes/yr (4×10^5 kg) of phosphorus for leaching to the groundwater and these are likely to end up eventually in the shallow lake systems or in Swan-Canning river system.

Some work on the nutrient contamination in groundwater beneath urban land use has been carried out (McFarlane 1984; Appleyard and Bawden 1986; Gerritse *et al.*, 1990) and some limited work has been carried out in quantifying N and P concentration in groundwaters directly beneath parkland sites (Barber *et al.*, 1991). These all indicate that NO_3 concentration is usually higher under Spearwood than under Bassendean Sands. The aquifer under Bassendean sands provides favourable conditions for denitrification (*i.e.*, high DOC, low pH and low redox potential). Because of a high iron content, the P adsorption is higher in Spearwood then Bassendean sands, therefore far more P leaching is expected through Bassendean sands. None of the studies have attempted to quantify nutrient leaching fluxes below the root zone and how this translates into the nutrient concentration of the groundwater. This is what is required if the eventual groundwater nutrient concentrations are to be predicted and managed. To be able to do this satisfactorily, both nutrient concentrations as well as water fluxes should be known in the unsaturated zone. Little quantitative information about these is available.

In recent years Perth residents have become some of the highest domestic consumers of water per head in Australian capital cities (WAWA 1985). For example average, per capita use of 321 m^3/yr in Perth compares to 167 m^3/yr in Sydney. Of the total household water used, about 50% is used exhouse, primarily for irrigation of lawns and gardens (WAWA 1985). Over 25% of households in Perth have private bores, which are used primarily for irrigation of lawns and gardens. Total exhouse water use by bore users (Scheme + groundwater from own bore) is estimated to be seven times the volume used by non-bore users. This is much higher than is required to maintain good quality lawns and gardens. Based on a survey, average actual irrigation application by non-bore users is less than the recommended value of 60% of net pan evaporation (Pan evaporation - rainfall) by WA Department of Agriculture. McFarlane (1984), based on a survey, reported that 80-90% of households apply water at a rate less than one net pan evaporation, while 70-80% apply at a rate of < 0.6 net pan evaporation. It is believed that very little or no deep drainage occurs when irrigation is applied at a net pan evaporation of < 0.6. But there are no hard data.

Only very limited information is available on the impact of urbanisation on groundwater recharge, and especially on recharge under components of land use in a household. It is suggested that urbanisation induces substantial increase in recharge. The majority of recharge (~70%) is contributed by roof and road runoff (McFarlane 1984). Septic tanks will also increase recharge. Overall recharge in urban Perth is estimated to be about 21% of the annual rainfall of 800 mm (Cargeeg *et al.*, 1987). In an unsewered area with a housing density of 10 ha^{-1}, a similar amount could be contributed through septic tanks (Whelan *et al.*, 1981).

How much recharge occurs under lawns and gardens has so far been a subject of speculation despite a few attempts to provide a quantitative answer. In the two urban catchments studied by McFarlane (1984) it was found that about 50% of the area is covered by gardens and lawns. Thus it is important to quantify accurately how much water and nutrients are passed below this area. This was broadly the aim of the present study.

1.1 Project Objectives

- To quantify water and nutrient (N, P) fluxes below the root zone of representative lawns on a number of urban sites.

- To evaluate the application of model(s) to predict transport of NO_3 and PO_4 through the unsaturated zone.

2. METHODS AND PROCEDURES

2.1 Climate and Soils

The experimental sites are located in the Perth metropolitan area (Fig. 1), on the Swan Coastal Plain, Western Australia. The area has a mediterranean climate with a rainy winter period from May to October and a dry summer period from November to March. The average rainfall of ~800 mm yr^{1}, is the primary source of recharge. Annually, potential evaporation (class A pan evaporation being ~1 800 mm yr^{-1}) far exceeds precipitation (Sharma and Huges 1985). During summer, irrigation is the primary source of water supply to lawns, and the amount of irrigation varies considerably (McFarlane 1984). Since the two major soil types, on the Swan Coastal Plain, Bassendean and Spearwood sands, (McArthur and Bettenay 1960) have different nutrient retention and possibly transformation characteristics, experimental sites were selected on both these soils. Experiments were conducted on two public parkland sites and two domestic lawn sites.

2.2 Site Selection

With the help of Water Authority of WA several public parks and domestic households were considered for selection. These were evaluated for representativeness, suitability in regard to access, availability of background data and owners agreeability. The selected domestic sites are located in Karrinyup (Spearwood sand) and Mount Lawley (Bassendean sand) as shown in the map (Fig. 1). Although the Mount Lawley site falls on the transition boundary of Bassendean/ Spearwood, it was still selected, as it more closely represented domestic lawn in a long established suburb. The two public parkland sites (Fig. 1) were selected based on availability of background information from an earlier study in which groundwater contamination of nutrients had been studied (Barber *et al.*, 1991). These sites were located at the Noranda Sporting Complex, Wyde Road, Noranda (Bassendean sand) and at Robinson Reserve, Royal Street, Tuart Hill (Spearwood sand). Important characteristics of the four selected sites and modes of operation/management for the sites are summarised in Table I.

2.3 Experimental Set Up

Within the selected lawn, a representative experimental site was selected, taking into consideration the minimal influence of trees, buildings, accessibility and protection of instruments.

All the sites selected are irrigated through automatic sprinkler systems. The water supply for three sites is from bores from the underlying shallow aquifer, while for the fourth site irrigation is derived from the public water supply (Table I).

Prior to selecting the exact location to install instruments for water and nutrient balance, a number of irrigation distribution tests were carried out so that the instruments could be placed at the most representative locations.

The water and nutrient fluxes passing below the root zone were measured by installing *in-situ* lysimeters, in which leachate was collected, pumped out and measured for water quantity and chemical composition. Two lysimeters were installed at each experimental site.

The components of input water (irrigation and precipitation) were measured separately. At least one water meter was installed in the water distribution line used for irrigation. A rain gauge was used at each site for precipitation measurement.

TABLE I.
CHARACTERISTICS AND MANAGEMENT PRACTICES OF SELECTED EXPERIMENTAL SITES

Name of Site	**Noranda**	**Tuart Hill**	**Karrinyup**	**Mount Lawley**
Type of site	Public	Public	Private	Private
Site use	Sporting Complex	Recreational Sporting Complex	Private Lawn	Private Lawn
Soil type	Bassendean	Spearwood	Spearwood	Bassendean
Year lawn established	1990	1980?	1984	1990
Maximum Rooting depth	50 cm	50 cm	50 cm	50 cm
Management of lawn clippings	Removed	Returned	Removed	Removed
Depth to water table	3 m	19 m	20 m	8 m
Source of irrigation	Bore	Bore	Mains	Bore
Depth of bore water extraction	30 m	36 m	-	12 m
Fertiliser type	Agran Mix	N/A	Cresco Lawn	Cresco Lawn
Application rate (kg/ha/yr)	1100	-	1750	1750
Fertiliser mix				
N	88	-	210	210
P	11	-	35	35
K	0	-	105	105

Changes in soil water storage were measured by the neutron moderation technique. Two neutron access tubes were installed to 1.5 m and readings were taken at 20 cm depth intervals. All the measurements were made on a weekly basis.

The input of nutrients (N, P) was quantified from all the possible sources. Fertiliser application rates and their exact composition were obtained from the Councils for the public parklands and from the home owners for the domestic lawns. During the study period, we applied the fertiliser on domestic lawns at a known and equivalent rate. The nutrient composition of irrigation water (from groundwater), precipitation and public water supply was determined on a number of occasions.

Two core (7.5 cm diameter) soil samples were taken at each site. The core samples were sectioned at 20 cm intervals down to 1.5 m depth, the final interval being 30 cm. Soil moisture content was determined. Soil solution extracts for the subsamples were prepared and chemically analysed for pH EC, phosphate, nitrate, sulphate and chloride. Total extractable phosphorus and iron were determined by extraction with 0.1 N H_2SO_4.

2.4 Chemical Analysis

The water samples were placed in a refrigerator during transport and stored in the laboratory before analysis. They were analysed within two weeks of collection by the Chemistry Laboratory, CSIRO Division of Water Resources, Perth.

Nitrate, NH_4, Cl, SO_4 and PO_4 were determined on a Technicon Auto Analyser using the following methods: NO_3 by a colorimetric Cadmium reduction technique (Technicon Industrial Method (TIM) No 158 - 71W/Preliminary); NH_4 colorimetrical; Cl colorimetrical by using ferric thiocyanate; PO_4 colorimetrical as reduced phospho-molybdate complex (TIM No. 329-74W/b). Total P was determined as PO_4 following digestion with a nitric and sulfuric acid mixture (APHA 1971). Dissolved OC was determined by infrared measurement of carbon dioxide following high temperature combustion.

2.5 Design and Installation of Lysimeters

Lysimeters of 40 cm x 40 cm x 30 cm dimension called mini-lysimeters were designed (Fig. 2) so that they could be installed in urban lawns, parklands and other land uses, where the land areas are not very extensive. The basic design of these lysimeters is similar to that of large lysimeters used in the rural horticultural areas (Sharma *et al.*, 1991), the main difference being in the method by which drainage is induced. The large lysimeters were free draining and therefore had to be installed far below the root zone (2 m). On the other hand, the mini-lysimeters are suction-drained, and therefore could be installed closer to the root zone (1.25 m). The mini-lysimeters are equipped at the bottom with a layer of fine-graded inert glass beads with sufficient air entry value that a suction of up to 50 cm H_2O could be applied either continuously or intermittently. This design was tested in the laboratory, a prototype was installed and tested beneath a representative lawn at the CSIRO grounds.

The lysimeters were constructed of 14 gauge galvanised iron. Seams were welded and treated with epoxy resin to prevent corrosion. The grid mesh support was secured in place by screws. Then a 25μm felt fabric was secured by neutral cure silicon sealant around the internal wall of the lysimeter. A 30 mm layer of 45-90 μm inert glass beads was placed on the felt fabric. A layer of 25 μm fabric was then placed on the glass beads and secured to the wall by neutral cure silicon sealant.

A procedure devised for installation of lysimeters involved repacking of original soil layers at the native bulk density and re-establishment of original turf. The top of the lysimeter was at least 1 m below the soil surface, being sufficiently below the turf's maximum rooting depth of 50 cm as usually encountered in the Perth metropolitan area. Associated equipment was installed in a control box so that a suction of 50 cm H_2O could be maintained in the lower chamber of the lysimeter. This was achieved by use of battery driven pump, a timer and a constant suction device. The design of the lysimeter and its layout along with associated parts are schematically shown in Figs. 2 and 3.

For lysimeter installation, intact lawn (lawn and soil) down to 20 cm was removed from an area of 60 cm x 70 cm and placed on a plastic sheet and covered. The soil was removed in 20 cm layers down to a depth of 1.5 m. Each soil layer was piled separately on a plastic sheet and kept covered. Sub-samples from each layer were obtained for chemical analysis and soil water measurement. The floor of the hole was then sloped slightly to ensure drainage of leachate to the outlet port of the lysimeter when installed.

Vacuum and leachate extraction tubes were then attached and sealed to the lysimeter ports. Soil from the bottom 30 cm layer was then placed into the lysimeter and compacted by hand using a steel compactor plate. The lysimeter was then lowered down into the hole and positioned. Soil from each layer was then replaced in 10 cm increments and compacted back evenly. Penetrometer readings were taken frequently to ensure that the correct compaction was obtained.

Finally, the intact lawn pieces were replaced and pressed down onto the soil. The vacuum and leachate tubes from the lysimeter were layed in a 10 cm x 20 cm trench leading several meters away to a control box.

2.6 Spatial Variability of Irrigation Input

Variability in the spatial distribution of irrigation water input was assessed at each site. This allowed computation of accurate irrigation inputs to each of the lysimeters, from the total irrigation applied over a given area. These measurements were required under a variety of weather conditions, in the light of the extent of measured spatial variability. Such measurements were necessary because at most sites, due to public access, irrigation input to lysimeters could not be measured directly. As an example, the pattern of spatial distribution of irrigation at the Noranda public park site is shown in Fig. 4.

2.7 Fertiliser Application and Management of Lawns

The public parks, Noranda and Tuart Hill were managed by the Shires of Bayswater and Stirling respectively. The domestic lawns were managed by the owners in their own way, except that fertiliser applications were based on the rate and time of application as recommended by the CRESCO fertiliser Co. The lawns were regularly mowed at 2 to 3 week intervals during summer and at a 4 to 6 week intervals during winter. Some of the important characteristics of the experimental sites are summarised in Table I.

3. RESULTS AND DISCUSSION

3.1 Hydrological Balance

3.1.1 Irrigation and Precipitation

At each site, the irrigation input from bores or mains was measured by metering the distribution pipe which supplies water to the site where the two lysimeters were located. This however only supplied the total volume of water on a given surface area. As discussed earlier, the portion of this total water falling on each of the lysimeters was estimated based on irrigation distribution patterns measured on one or two occasions at each site.

In most cases the differences in irrigation input between the two lysimeters at a site were less than 10%, only at one site (Noranda) were the differences greater, up to 18%.

Precipitation input at three sites was measured using one rain gauge, and considered to be the same for the two lysimeters. The precipitation input for the unmeasured site was assumed to be the same as that measured on a site close by where precipitation was measured.

Thus total water input (I_t) for each lysimeter was a sum of irrigation (I) and precipitation (P).

The measurement period was 210 days (5/12/91 through to 25/6/92) for three sites, while it was 177 days for the fourth site, Tuart Hill (30/12/91 through to 25/6/92). Over the entire period, average water input rates for the four sites ranged from 3.84 mm/day (Karrinyup site) to 5.17 mm/day (Mt Lawley site).

For convenience, the entire observation period was divided into two hydrological periods:

(i) Irrigation-dominated period (until 21 May 1992); and
(ii) Precipitation-dominated period (21 May 1992 to 25 June 1992).

TABLE II.
SUMMARY OF THE COMPONENTS OF WATER BALANCE FOR FOUR URBAN EXPERIMENTAL SITES.

Site	Period	$\bar{R}$ (mm)	$\bar{I}_T$ (mm)	$\bar{I}$ (mm)	$\bar{P}$ (mm)	$\bar{E}_T$ (est) (mm)	$\bar{E}_p$ (mm)	$\bar{R}/\bar{I}_T$	$\bar{I}/\bar{I}_T$	$\bar{I}_T/\bar{E}_p$
Tuart Hill	I	330	672	446	226	342	721	0.491	0.664	0.932
	II	126	165	9	156	39	84	0.764	0.055	1.964
	Total	456	837	455	382	381	805	0.545	0.544	1.040
Noranda	I	121	759	503	256	638	919	0.159	0.663	0.826
	II	197	262	62	200	65	84	0.752	0.237	3.119
	Total	318	1021	565	456	703	1003	0.312	0.553	1.018
Karrinyup	I	270	592	329	263	322	919	0.456	0.556	0.644
	II	154	249	88	161	95	84	0.619	0.353	2.964
	Total	424	841	417	424	417	1003	0.504	0.496	0.839
Mount Lawley	I	688	1020	758	262	332	919	0.675	0.743	1.110
	II	139	162	0	162	23	84	0.858	0	1.929
	Total	827	1182	758	424	355	1003	0.700	0.641	1.179

During the period until 21/5/92, when irrigation was the dominant input, irrigation amounted to over 66% of the total input for the three sites, and some 53% of the input for the Karrinyup site (Table II). The proportional irrigation input would have been even higher except for unusually heavy rain (125 mm) which fell over a 2 day period. This event contributed at least 25% of the total water input during the summer period. Disregarding this rainfall during the summer period, irrigation accounted for 95 to 100% of the water input. During June the contribution of irrigation varied from 0 to 33% (Table II).

3.1.2 Soil Water Storage

Soil water storage measured for the 1.5 m profile over a period of one month showed that the changes in water storage were not more than 30 mm at all the sites. This is because the soils are kept at high water content through irrigation. Therefore, for calculating long term water balance over the six month period, the changes in soil water storage could be considered negligible.

It is anticipated that for short periods during summer (weekly or fortnightly) changes in water storage could be of much larger magnitude and therefore, when calculating water balance for short periods, such changes need to be taken into account (Sharma *et al.*, 1991).

3.1.3 Deep Drainage and Water Balance

The weekly water fluxes (recharge) measured by lysimeters at each of the four experimental sites, the corresponding cumulative average recharge and water input are presented in Figs. 5 through 12.

Average hydrological parameters, *i.e.*, recharge ($\overline{R}$), total water input ($\overline{I_T}$), irrigation input ($\overline{I}$), precipitation input ($\overline{P}$) and pan evaporation ($\overline{E_p}$) for the two hydrological periods as discussed earlier and for the entire period are summarised in Table II. Also included in this table are some of the ratios of selected parameters and estimated values of average evapotranspiration ($\bar{E}_T$). Evaporation computations assume that there is no surface runoff (*i.e.,* all I_T infiltrates into the profile) and soil water storage changes are negligible. Salient water balance features for each of the four sites are described below:

Noranda

Reproducibility of results from the two lysimeters is excellent. One lysimeter is consistently higher (within 10%) but this is consistent with the total water input (Figs. 5 and 6).

However, it should be noted that despite similar amounts of irrigation input throughout the summer, recharge is very low. During the summer period, $\bar{R}$ is only 0.16 of $\bar{I}_T$, which is 1/3 to 1/5 of the other sites (Table III). During the winter period, $\bar{R}$ is much higher proportion (0.75 $\bar{I}_T$), which is consistent with the results at other sites.

Furthermore, examination of data shows that an exceptional rainfall event of about 125 mm/week during summer (6/2 - 13/2/92) produced only 1/3 to 1/5 of the recharge measured at the other three sites. Considering similarity in soil type and grass condition, such differences are not possible unless input water was being lost through some other process. Computations show that differences in changes in soil water storage cannot account for such large differences.

Approximate computations (Table II) show that approximate estimates of $\bar{E}_T$ (total water input minus headage) for this site would be about double that of a comparable parkland site (Tuart Hill).

Considering all these features and based on some observations during irrigation periods, it is concluded that during summer due to the hydrophobic nature of dry surface sands a considerable proportion of applied irrigation water runs off on a localised scale and does not infiltrate. This water repellency property of Bassendean sands has been reported earlier (Roberts and Carbon 1971). The magnitude of repellency depends among other factors, on the degree of wetness of the surface soil. The amount of surface runoff will vary during irrigation periods. Once the surface soil is wetted and kept at a high water content, the water repellency effects diminish or completely disappear. This appears to be the case during winter periods.

It must be emphasised that while localised recharge is reduced due to water repellency, it is unlikely that the overall recharge for the entire oval (in the case of the Noranda Sporting Complex) would be affected significantly, as surface runoff from one area will infiltrate into another and produce extra recharge. The overall effect on recharge is not easily quantifiable.

Tuart Hill

Reproducibility of the two lysimeters is excellent. In general lysimeter outputs are consistent with inputs. Overall the results are consistent with what would be expected. (Figs. 7 and 8).

Mount Lawley

The reproducibility of the two lysimeters is good. Recharge from lysimeter 1 is consistently higher than lysimeter 2, but these results are consistent with the water input values. Initial irrigation input was underestimated by about 15% and it has now been corrected (Figs. 9 and 10).

Karrinyup

Recharge from the two lysimeters is extremely different (Figs. 11 and 12). Although the water inputs for the lysimeters are similar throughout the period, the recharge is consistently two to three times higher in lysimeter 1 than in lysimeter 2. During the week of the extreme rainfall event in summer of 125 mm, there was a three fold difference in recharge between the two lysimeters. Recharge in lysimeter 2 was much closer to that measured at other sites during similar period. From further investigations, it was revealed that water was leaking out of lysimeter 1 and therefore lysimeter 1 results were abandoned.

3.1.4 Overall Comments

The water balance results show that during summer irrigation accounted for over 90% of the total water input, except for the week of the extreme rainfall event. During the winter period after the onset of rain, irrigation input was reduced or completely eliminated (Table II).

Irrigation as a proportion of total input $(\bar{I}_T)$ was highest for the Mt Lawley Site. The order of sites for $\bar{I}_T$, and $\overline{I}/\overline{I}_T$ was the same, and it was: Mt Lawley > Noranda > Tuart Hill > Karrinyup.

However, the order of sites in terms of total recharge, as well as R/It for the entire measurement period was: Mt Lawley > Tuart Hill > Karrinyup > Noranda. Thus, recharge was roughly proportional to water in put irrespective of soil type as well as lawn management. The measured recharge as a fraction of total water input ranged from 0.50 to 0.76 for the three sites. An exception to this was the Noranda site, possible reasons for which have been discussed earlier.

The total water input for the entire period as a fraction of pan evaporation ranged from 0.80 to 1.08. During summer, for most sites this ratio was close to one except for the extreme event during February (Table II, Figs. 5, 7, 9, 11). This is in contrast to some earlier survey observations that most households apply water at a rate of < 0.6 net pan evaporation (McFarlane 1984).

Although not much information was available on exact recharge values under lawns for the Perth area. Recharge was noted as being > 50% of total input in this study and this is much higher than reported earlier (McFarlane 1984). These results compare reasonably well with those for a winter grassland on the Swan Coastal Plain under rain-fed conditions (Sharma *et al.*, 1989).

3.2 Water Quality and Nutrient Balance

3.2.1 Nutrient Input

The public park sites are managed by the two respective Shires and fertiliser rates are those normally used by the Shires (Table I). According to the relevant Shire the Tuart Hill site did not receive any fertiliser for at least the previous five years.

In the case of the two domestic lawns, the fertiliser inputs adopted were those recommended by a fertiliser company, and the owners agreed to apply at these rates. These were 210 kg/ha N, 35 kg/ha P and 105 kg/ha K. We were advised that previous fertiliser applications were of similar magnitude.

Other possible sources for nutrients are irrigation and rainfall. Table III shows that there was significant input of nitrogen through irrigation water at Tuart Hill and Mt Lawley and some at Noranda. The Noranda site being in Bassendean sand, there is likelihood of denitrification in the groundwater (Gerritse *et al.*, 1990; Barber *et al.*, 1991). As the Karrinyup site is irrigated from the mains water supply, the input of N is rather small (~.025 kg/ha). The composition of chloride in bore water was similar to that from the mains water supply (~100 mg/l).

TABLE III.
AVERAGE CHEMICAL COMPOSITION OF PRECIPITATION AND BORE WATER (USED FOR IRRIGATION) AT URBAN EXPERIMENTAL SITES.

Site	**Precipitation**		**Bore Water**			
	Cl (mg/l)	**pH**	**NO_3-N (mg/l)**	**PO_4-P (mg/l)**	**Cl (mg/l)**	**pH**
Tuart Hill	9.70	8.26	6.80	0.003	116	7.03
Noranda	10.45	7.54	0.084	< 0.002	129	6.49
Mount Lawley	8.97	7.67	3.63	< 0.002	112	6.38
Karrinyup	8.12	7.99	Bore water not used			
Overall Average	9.30	7.86				

While there was some input of chloride through precipitation (Table IV), there is usually negligible input of N and P (Sharma *et al.*, 1991).

3.2.2 Chemical Composition of Drainage Water

Concentrations and yields in leachates of chloride for the four experimental sites are presented in Figs. 13-20, of NO_3-N in Figs. 21-28 and of PO_4-P in Figs. 29-35. It should be noted that concentrations are given only for one lysimeter at the Karrinyup site as the results from the other lysimeter were unreliable.

The average values of input and output of Cl, nitrogen (NO_3-N and NH_4-N) and PO_4-P, calculated based on the irrigation-dominated period (until 21/5/92) and precipitation dominated period (31/5/92 to 25/5/92) and also for the entire period are summarised in Tables IV, V, and VI respectively.

3.2.3 Chloride Balance

There is a seasonal variation in the Cl concentration of leachate, ranging usually from about 100 mg/l to 300 mg/l. An exception to this is the Noranda site at which Cl concentrations of up to 600 mg/l were measured. The total yield of Cl over the 210 day period was highest for Mt Lawley, while the other three sites were comparable (Table IV).

Chloride, being a conservative tracer, is assumed to move with water. The primary source of Cl is irrigation water. The Cl concentration in groundwater was relatively uniform, (ranging from 116 to 130 mg/l), which was some what similar to that in mains water ($\sim$100 mg/l). At two sites, fertilisers added substantial amounts of Cl (Table IV). Precipitation contributed 5-20% of the total chloride input.

Using a steady state chloride balance method (Sharma *et al.*, 1989), recharge as a fraction of total water input for the four sites was: 0.25 (Noranda), 0.4 (Tuart Hill), 0.53 (Mt Lawley) and 0.25 (Karrinyup). These estimates ignore chloride storage changes in the soil profile.

TABLE IV.
SUMMARY OF COMPONENTS OF CHLORIDE BALANCE FOR FOUR URBAN EXPERIMENTAL SITES.

Site	**Period**	**Output**		**Total Input**	**Components of Input**					$\bar{C}_{NO3\text{-}N}/\bar{C}_{Cl}$ *
					I		**P**		**Fertiliser**	
		kg/ha	$\bar{C}_R$ **mg/l**	**kg/ha**	**kg/ha**	$\bar{C}_I$ **mg/l**	**kg/ha**	$\bar{C}_p$ **mg/l**	**kg/ha**	
Tuart Hill	I	541	164	540	518	116	21.9	9.70	0	2.88×10^{-2}
	II	216	171	25	10	111	15.2	9.70	0	1.62×10^{-2}
	Total	757	166	565	528	116	37.1	9.70	0	2.52×10^{-2}
Noranda	I	350	289	681	654	130	26.8	10.45	0	6.77×10^{-2}
	II	590	299	102	81	131	20.9	10.45	0	1.54×10^{-2}
	TOTAL	940	296	783	735	130	47.7	10.45	0	3.48×10^{-2}
Karrinyup	I	548	203	401	272	83	23.6	8.97	105	1.50×10^{-2}
	II	642	417	95	73	83	14.4	8.97	0	0.46×10^{-2}
	TOTAL	1190	281	488	345	83	38.0	8.97	105	0.92×10^{-2}
Mount Lawley	I	929	135	975	849	112	21.3	8.12	105	8.79×10^{-2}
	II	257	185	13	0	-	13.1	8.12	0	0.85×10^{-2}
	TOTAL	1186	143	988	849	112	34.4	8.12	105	7.05×10^{-2}

* Ratio of flow-weighted concentration of NO_3-N and Cl in the leachate.

Overall, recharge estimates based on chloride were somewhat lower than those measured (Table II). More accurate information on chloride input for each irrigation and precipitation event should improve the accuracy of such estimations.

3.2.4Nitrogen Balance

At the three sites where fertiliser was applied, fertiliser-N accounted for most (90-100%) of the total applied N. In the case of Tuart Hill, where no fertiliser was applied, approximately 30 kg/ha nitrogen was applied through irrigation over the observation period. Total N application at the other sites ranged from 80 to 235 kg/ha over the experimental period.

Nitrogen leaching was measured in the form of NO_3-N and NH_4-N. The NH_4-N component accounted for ~2% (Noranda) to 19% (Karrinyup) of the total N output (Table V).

There was a strong seasonal variation of NO_3-N concentration in the leachate, and there was a strong variability among sites (Figs. 21-28). The NO_3 peaks in leachate reflected the influence of the timing of N application through fertiliser. Far less seasonal variation of NO_3-N in leachate at Tuart Hill may be because the source of N is irrigation water, which is applied uniformly over the summer period. Overall the two Bassendean sites (Noranda and Mt Lawley) show much higher NO_3-N concentrations than the Spearwood sites (Tuart Hill and Karrinyup). It is interesting to note that a heavy rainfall event in mid February caused a strong peak in NO_3-N concentration at most sites, although the magnitude of the peak and its duration were quite different among sites.

NO_3-N concentrations for the Spearwood sites (Figs. 24, 27) ranged from 1 mg/l to 10 mg/l, with most being between 2 to 6 mg/l. But for the Bassendean sites (Figs. 22, 25), NO_3-N concentrations ranged from < 1 to 30 mg/l (Mt Lawley), and < 1 to 45 mg/l (Noranda). It should be noted that for the Mt Lawley and Karrinyup sites, the fertiliser application was the same.

TABLE V.
SUMMARY OF COMPONENTS OF NITROGEN BALANCE FOR FOUR URBAN EXPERIMENTAL SITES.

Site	Period	Total Output		Total Input	Components of Input			Components of Output				$\frac{NO_3\text{-}N}{N_{tot}}$	$\frac{NH_4\text{-}N}{N_{tot}}$
								NO_3-N		NH_4-N			
		kg/ha	$\bar{C}_R$ mg/l	kg/ha	I kg/ha	$\bar{C}_I$ mg/l	Fertiliser kg/ha	kg/ha	$\bar{C}_R$ mg/l	kg/ha	$\bar{C}_R$ mg/l		
Tuart Hill	I	17.2	5.21	30.4	30.4	6.82	0	15.6	4.73	1.6	0.49	0.91	0.09
	II	3.6	2.86	0.5	0.5	5.56	0	3.5	2.78	0.1	0.08	0.97	0.03
	TOTAL	20.8	4.56	30.9	30.9	6.79	0	19.1	4.19	1.7	0.37	0.92	0.08
Noranda	I	23.9	19.80	80.4	0.42	0.084	80	23.6	19.50	0.3	0.25	0.99	0.01
	II	9.3	4.72	0.1	0.05	0.084	0	9.1	4.62	0.2	0.10	0.98	0.02
	TOTAL	33.2	10.44	80.5	0.47	0.084	80	32.7	10.28	0.5	0.16	0.98	0.02
Karrinyup	I	7.0	2.59	210.1	0.06	0.02	210	5.1	1.89	1.9	0.70	0.73	0.27
	II	2.1	1.36	0.0	0.01	0.01	0	1.1	0.71	0.3	0.19	0.52	0.14
	TOTAL	8.4	1.98	210.1	0.07	0.02	210	6.2	1.46	2.2	0.52	0.74	0.26
Mount Lawley	I	81.3	11.82	232.7	22.7	2.99	210	80.9	11.76	0.4	0.06	1.00	0.00
	II	2.3	1.65	0.0	0.0	-	0	2.1	1.51	0.2	0.14	0.91	0.09
	TOTAL	83.6	10.11	232.7	22.7	3.99	210	83.0	10.04	0.6	0.07	0.99	0.01

$\bar{C}_R$ = flow-weighted mean concentration in recharge water
$\bar{C}_I$ = flow-weighted mean concentration in irrigation water
N_{tot} = Total N output (NO_3-N + NH_4-N)

The flow-weighted NO_3-N concentration in leachate for the four sites ranked as Karrinyup < Tuart Hill < Mt Lawley < Noranda. The respective concentrations were 1.46, 4.20, 10.04 and 10.28 mg/l (Table V). As commented earlier, the accepted NO_3-N concentration limit for drinking water is ~10 mg/l. Thus it is clear that on many occasions, leachate below lawns contained NO_3-N concentrations which are much higher than the acceptable drinking limit.

The NO_3-N yield for the six-month period for Tuart Hill (~19 kg/ha) and Karrinyup (~7 kg/ha) contrasted with that of Noranda (33 kg/ha), and Mt Lawley (83 kg/ha) as shown in Figs. 22, 24, 26, 28.

Accepting that no N is applied through fertiliser at the Tuart Hill site, N inadvertently applied through irrigation water is sufficient for the maintenance of a healthy lawn of the indigenous type (Kikuyu). About 60% of N applied through irrigation is leached and returned to the aquifer.

At the two domestic lawn sites supplied with fertiliser, from 4 to 40% of the applied nitrogen is leached, primarily as NO_3-N (Table V). In heavily fertilised (at a rate of > 1 500 kg/ha N) horticultural sites, leachate accounted for about 40% of the total applied N (Sharma *et al.*, 1991).

3.2.5 Phosphorus Balance

The primary source of P was fertiliser, as P concentrations in irrigation and precipitation are negligible.

Overall PO_4-P concentrations in leachate were low. The pattern of leaching at the four sites was highly variable and complex. For example, at Noranda PO_4-P concentration was usually > 0.1 mg/l, with peaks reaching close to 1 mg/l. At Tuart Hill, where no P is being applied, PO_4-P concentrations were usually < 0.1, but there were peaks up to 2.5 mg/l. In the case of the two domestic sites being fertilised currently, PO_4-P concentrations were mostly < 0.1 mg/l but peaked to 0.4 mg/l.

TABLE VI.
SUMMARY OF COMPONENTS OF PHOSPHORUS BALANCE FOR FOUR URBAN EXPERIMENTAL SITES.

Site	Period	Output		Input	Components of Input		
					I		Fertiliser kg/ha
		kg/ha	$\bar{C}_R$ mg/l	kg/ha	kg/ha	$\bar{C}_I$ mg/l	
Tuart Hill	I	0.33	0.10	0.02	0.02	0.004	0
	II	0.24	0.19	0	0	-	0
	TOTAL	0.57	0.125	0.02	0.02	0.004	0
Noranda	I	0.13	0.107	7.50	Negligible		7.5
	II	0.30	0.152	0	Negligible		0.0
	TOTAL	0.43	0.135	7.50	Negligible		7.5
Karrinyup	I	0.05	0.019	35	Negligible		35
	II	0.06	0.039	0	Negligible		0
	TOTAL	0.11	0.026	35	Negligible		35
Mount Lawley	I	0.15	0.022	35.02	0.02	0.003	35
	II	0.03	0.022	0	0	-	0
	TOTAL	0.18	0.022	35.02	0.02	0.003	35

Based on the flow-weighted PO_4-P concentrations in leachate for the entire period, the order of sites was: Mt Lawley < Karrinyup < Tuart Hill < Noranda, and the respective concentrations were 0.022, 0.026, 0.125 and 0.135 mg/l.

The total PO_4-P output in recharge ranged from 0.11 kg/ha to 0.6 kg/ha, a small proportion of the applied P. It appears that soils at all the three sites, where fertiliser is being applied, are still adsorbing P. However, it appears that at Tuart Hill previously adsorbed P is being released from the soil and leached.

3.2.6 General Discussion

Despite some attempts, quantitative estimates of the proportion of water input which becomes recharge under urban irrigated lawns has been a subject of conjecture, especially in the Perth region (*e.g.*, Williamson and Cole 1976; McFarlane 1984). Estimates based on water balance studies vary from 10% to 22% of the water input.

There has been acceptance that if irrigation was applied at a rate of 0.6 net per evaporation (Ep), recharge would be negligible (McFarlane 1984). In some other cities the recommended rates may be even lower (Moore 1977). Obviously, in addition to total water application, the intensity and frequency of irrigation would be important factors in this regard. Within the Perth metropolitan area, when lawns are irrigated from bores, the amount of irrigation applied is likely to be much higher than 0.6 Ep. This applies to about 25% households and most public parks and gardens maintained by local Shires (WAWA 1985). Our results show that in a domestic site with private bore (Mt Lawley), about 70% of the irrigation returned as recharge, while in a public park site (Tuart Hill) about 50% of the applied returned to the aquifer. In the case of one public site (Noranda), due to water repellency, the measured recharge was only 16% of the irrigation, while the overall areal recharge is expected to be much higher. Even in a domestic site (Karrinyup) which is irrigated from the mains, a typical site without bore, about 40% of the applied irrigation passes below the root zone. Our data show that overall, there is excessive irrigation in urban areas, and this is likely to cause leaching of nutrients. Such results are not unusual, as about 50% of the water input has been reported to contribute to recharge on irrigated turfs in the USA (Morton *et al.*, 1988).

There are numerous papers reporting increased leaching of solutes, especially NO_3-N, with increased irrigation for turf grasses (Snyder *et al.*, 1984; Morton *et al.*, 1988) as well as crops (*e.g.*, Timmons and Dylla 1981; Herbert 1986). Our results for the two domestic lawns are in accord with this general theme. Higher irrigation rates have obviously induced increased leaching of NO_3 at Mt Lawley. Leachate beneath this site contains a flow-weighted mean concentration of NO_3-N which exceeds the WHO's drinking water limit of 10 mg/l. The leachate beneath a public park site (Noranda) also contains such high NO_3-N concentration especially during summer, possibly because of water loss through surface runoff due to water repellency. Leachates at all the sites however, contain sufficiently large NO_3 concentrations to be unacceptable from the view point of coastal estuaries and wetlands (Ryther and Dunstan 1971; Vollemweiller 1985; Ryding and Rast 1989; Davis *et al.*, 1991). The NO_3 concentrations observed in leachate are likely to be lower than those in the groundwater, primarily due to dilution and in some cases (*e.g.*, Bassendean sands) due to denitrification (Gerritse *et al.*, 1990; Pionke *et al.*, 1990; Barber *et al.*, 1991; Sharma *et al.*, 1991). This is supported by the NO_3-N concentrations in the bore irrigation waters (Table V), being 3.36 mg/l and 6.81 mg/l for the two Spearwood sands, and 0.084 mg/l for the Bassendean sand (Noranda).

Differences in the pattern of NO_3-N concentration between the two domestic sites (Mt Lawley and Karrinyup), which received the same fertiliser and application rates, can be ascribed to differences in irrigation regime, soil type and extra N input through irrigation. The soil at Karrinyup site has a slightly higher water holding capacity because of a higher clay fraction with higher iron content. These properties are likely to cause reduced N and P leaching at this site.

The overall PO_4-P concentrations in leachate at the two public park sites are higher than the domestic sites. This is despite the fact that P application rates are much higher at the domestic sites. The differences in PO_4-P concentrations between the two domestic sites, as well as between the two public sites, are rather small. Overall P concentrations observed beneath these sites are comparable with those found in the groundwaters on the Swan Coastal Plain (Gerritse *et al.*, 1990; Pionke *et al.*, 1990; Sharma *et al.*, 1991; Barber *et al.*, 1991). It is clear, as confirmed by the depth distributions of PO_4-P and total P in the soil profile that the main phosphorus front is still well within the top 1.5 m of the soil profiles and that it will take several years to reach the groundwater (Sharma *et al.*, 1991). It is obvious that some phosphorus ($<$ 2% of applied P) is still leaching through the soil system. It appears that this leaching is enhanced at high flow rates (Figs. 29, 31, 33, 35).

4. MODELLING NUTRIENT TRANSPORT THROUGH SOIL PROFILES

4.1 Modelling Approach

We surveyed the literature in regard to various approaches to model nutrient leaching through the soil profile. Models range from being deterministic/mechanistic to purely empirical (Vachaud *et al.*, 1990). Between these lie some models which are based on the conservation of mass but which may not consider the various mechanisms involved. We have been developing and adapting mechanistic as well as simplified mass balance models.

To model chemical transport under steady water flow conditions, we have investigated the application of a one-dimensional convective/dispersive transport equation (Van Genuchten 1977; Van Genuchten and Gray 1978) *i.e.*,

$$D\frac{\partial^2 c}{\partial z^2} - v\frac{\partial c}{\partial z} - R\frac{\partial c}{\partial t} = \mu c - \gamma \tag{1}$$

D = apparent dispersion coefficient [L^2T^{-1}], c = solute concentrations in the soil solution [ML^{-3}], z = vertical coordinate [L], t = time [T], v = pore water velocity [LT^{-1}] (= q/θ where, q = constant water flux [LT^{-1}] and θ = volumetric soil moisture content [L^3L^{-3}]), μ and γ = rate coefficients for first order decay and zero order production respectively [T^{-1}] [$ML^{-3}T^{-1}$], and R = retardation factor [1]. Retardation factor for a solute with a linear adsorption isotherm (s = kc, where s = concentration of solute in the solid phase [MM^{-1}], and k = empirical distribution constant [$M^{-1}L^3$]) is given by:

$$R = 1 + \frac{\rho k}{\theta} \qquad (2)$$

where ρ = soil bulk density [ML^{-3}].

For intensively irrigated sandy soils the upper boundary condition could be approximated by the flux type boundary condition (Sharma *et al.*, 1991b) *i.e.*,

$$\left[-D \frac{\partial c(z,t)}{\partial z} + vc(z,t) \right]_{x=0^{+}} = vC_i(t) \qquad (3)$$

where $C_i(t)$ = concentration of solute applied to the soil surface [ML^{-3}]. The initial condition $C_p(x,t=0)$ [ML^{-3}] is given by a concentration profile of solute obtained at the beginning of the modelling period.

4.1.1 Quasi-Steady State Model

Field systems with uniform, diffuse irrigation supplemented by occasional precipitation and sporadic application of fertilisers could be approximated by steady state conditions for short periods of time (1-3 weeks). Within each period $\Delta\tau$ (assumed to be a week), volumetric soil moisture content, water flux, water pore velocity, input concentration of the solute, the dispersion coefficient, the retardation factor and the sink rate coefficients could be considered constant and approximated by average weekly values (*i.e.*, $\bar{\theta}(\tau_n)$, $\bar{q}(\tau_n)$, $\bar{v}(\tau_n)$, $\bar{C}_i(\tau_n)$, $\bar{D}(\tau_n)$, $\bar{R}(\tau_n)$, $\bar{\mu}(\tau_n)$ and $\bar{\gamma}(\tau_n)$ respectively, where $\tau_n = n\Delta\tau$, n = 0, 1, ...) obtained from field data.

In the quasi-steady state approach, the transport equation (1) is solved for each week with constant parameters given by the average values. Initial concentration profile $C_p(x,\tau_{n+1})$ for each week is taken as a final profile $c(x,t=\tau_n)$ modelled for the preceding week.

The quasi-steady state transport model and the multi-dimensional inverse parameter estimation technique were implemented numerically in FORTRAN 77.

4.1.2 Model Calibration and Testing

Acquisition of realistic values of parameters (D, v, R, μ, γ) in the transport equation (1) is crucial for effective modelling of nutrient transport and transformations under field conditions. Some of the parameters could be obtained from water and nutrient transport in soil columns in controlled laboratory experiments. This approach often leads to implausible estimates mostly due to the spatial variability of hydro-chemical properties within the field (Sharma and Luxmore 1979), but also due to unavoidable disturbance of the soil structure during preparation of laboratory experiments.

We applied inverse parameter estimation techniques to calibrate the model. Several forms of the objective function (deviations between lysimeter concentration data and model predicted output) were formulated and tested in order to minimise instability and avoid non-uniqueness of estimated parameters (Kool *et al.*, 1987). Multi-dimensional optimisation algorithms were employed to minimise the suitable objective function.

Initially the calibrated model was tested for a cauliflower crop grown on a horticultural site for which components of water and solute (NO_3, NH_4, PO_4, Cl) had been quantified using lysimetry and other measurements (Sharma *et al.*, 1991b).

The inverse parameter estimation technique was applied to calculate the dispersion coefficient function from measured Cl concentrations in the leachate. A chloride concentration profile obtained from core samples taken before commencement of the crop period was used as the initial condition. The upper boundary condition was assumed to be of a flux type and given by the average chloride concentration in irrigation water. Figure 37 shows a close agreement between Cl concentration measured by a lysimeter and concentration predicted by the model at 2.0 m depth.

The model was then applied to simulate nitrogen transport. Using the previously obtained dispersion function (using Cl data) and estimated input concentration of mineralisable N, the zero order production rate coefficient was calculated through inverse optimisation using measured concentrations of NO_3-N in leachate. Figures 38 and 39 show the NO_3-N concentration of leachate simulated with and without production terms respectively. It can be seen that substantial NO_3-N was derived from the soil organic pool during the season. A comparison of modelled NO_3-N concentration with the measured values shows very good agreement.

4.2 Modelling Cl, NH_4 and NO_3 Leaching under Lawns

The quasi-steady state model was applied to simulate Cl, NH_4-N and NO_3-N leaching below the root zone for a period of 168 days (09/01/92 - 25/06/92) at two urban lawn sites (Tuart Hill and Mount Lawley).

Chloride was taken as a conservative tracer and its uptake by plants was assumed negligible. Both nitrogen species (NH_4 and NO_3) were also treated as conservative chemicals (adsorption on soil particles was assumed negligible). This is especially true for nitrate below the root zone in the vadose zone of well drained sandy soils, where aeration is sufficient and denitrification is expected to be negligible. Water content for sandy soils in the vadose zone usually does not exceed 0.10 $cm^3\ cm^{-3}$ (when θ_s = 0.30 $cm^3\ cm^{-3}$).

For both sites, the average weekly pore-water velocity $\bar{v}(\tau_n)$ was calculated from measurements of weekly recharge, total water input (irrigation and precipitation), evapotranspiration (estimated) and soil water content $\bar{\theta}(\tau_n)$. Input concentration $\bar{C}_i(\tau_n)$ was calculated from the net water input (= total water input - evapotranspiration) and from the mass of applied nutrients. Provision was made for situations where total water input was equal to or greater than evapotranspiration. We assumed that the chemical was stored in the surface layer and dissolved in following weeks. Initial concentration profile $C_p(x,\tau_n)$ (for Cl, NH_4-N and NO_3-N) was obtained from core samples taken at the beginning of the modelling period.

4.2.1 Simulating Cl Transport

The quasi-steady state model was applied to simulate chloride transport through the soil profile for the Tuart Hill site. The retardation factor was assumed to be one, production and decay rate coefficients were zero (Eqn. 1). We also made an assumption, that the dispersion was a linear function of the pore-water velocity *i.e.*,

$$\bar{D}(\tau_n) = \alpha \bar{v}(\tau_n) \tag{4}$$

where α = constant coefficient [L].

The coefficient α was computed through the inverse parameter estimation technique (Sharma *et al.*, 1991). The objective function $O(\alpha)$ that was minimised was expressed as the ordinary least-squares (OSL) formulation for chloride concentrations in the leachate and model response *i.e.*,

$$O(\alpha) = \sum_{n=0}^{m} [\bar{C}_{lys}(\tau_n) - C_\alpha(\tau_n)]^2 \tag{5}$$

where $\bar{C}_{lys}(\tau_n)$ = concentration of chloride in the leachate for the n-th week and $C_\alpha(\tau_n)$ = model response at the lysimeter level for a given parameter α. The predicted value of α = 0.0521 m was well within the expected range for the Spearwood sand. Figure 40 shows concentration of chloride measured by the lysimeter and model simulation for the optimised parameter α. The dispersion function $\bar{D}(\tau_n)$, calculated from Eqn 4, is also shown.

Next, we applied the model to predict chloride leaching for the Mount Lawley site. Using the coefficient α obtained for the Tuart Hill site and from the average weekly pore-water velocity $\bar{v}(\tau_n)$, we calculated the dispersion function $\bar{D}(\tau_n)$. A comparison of chloride concentrations at the lysimeter level predicted by the model with measured values (Fig. 41) show a very good agreement.

4.2.2 Modelling Nitrogen (NH_4 and NO_3) Transport

In predicting the transport of NH_4 through the unsaturated zone all components of the mass balance should be considered. At the Mount Lawley site, virtually all NH_4 applied to the soil surface was in a form of mineral fertiliser (NH_4 from irrigation < 0.3%). Both volatilisation and plant uptake of NH_4 could be considered negligible. Two dominant processes (mineralisation and nitrification) within the system (root zone) were responsible for transformations of N-species relevant to NH_4 transport (Jansson *et al.*, 1982; Parton and Porter 1985). Some NH_4 was produced from nitrogen contained in organic matter through mineralisation, while most NH_4 was converted to NO_3 through nitrification. The nitrification process, being concentration dependent, could be represented by the first order decay rate coefficient $\bar{\mu}_{NH_4\text{-}N}$, while mineralisation, being concentration independent, could be represented by the zero order production rate coefficient $\bar{\gamma}_{NH_4\text{-}N}$ (Eqn. 1).

We applied the inverse parameter estimation technique to the NH_4-N concentrations measured by the lysimeter and simultaneously calculated the rate coefficients $\bar{\mu}_{NH_4\text{-}N}$ and $\bar{\gamma}_{NH_4\text{-}N}$. We used the same dispersion function $\bar{D}(\tau_n)$ as for the chloride simulations. Then, with the estimated rate coefficients, we used the model again and computed concentration profiles of decayed NH_4-N for every time step t_i. Since nitrification of NH_4-N is almost solely responsible for the decay, it was assumed that the NH_4-N decay profiles numerically represented the increase in nitrate concentration $\Delta c_{NO_3\text{-}N}(x,t_i)$ in the system.

The next phase of modelling N species at the Mount Lawley site involved simulations of NO_3-N transport through the soil profile. Again, the same dispersion function desired for Cl simulations was used. We considered that apart from leaching, plant uptake and immobilisation use the dominant processes of removing nitrate from the system (denitrification is likely to be negligible).

During execution of the model the nitrate that originated from nitrification was combined with the modelled concentration *i.e.*, the concentration profile $c_{NO_3\text{-}N}(x,t_i)$ was adjusted by $\Delta c_{NO_3\text{-}N}(x,t_i)$ before the concentration profile for the following time step t_{i+1} was calculated.

Resulted NO_3-N concentrations simulated by the model at the lysimeter level are shown in Figure 42 and compare well with the field results.

The model simulations compare well with lysimeter data. Figure 43 compares nitrate-N yield computed from the model simulation with the lysimeter data. The total deviation for the modelled period (168 days) was about 8%.

4.3 Modelling Phosphate Leaching

For a highly sorbing solute (*e.g.*, PO_4), the terms D, μ and γ in Eqn. (1) can be ignored. A non-linear time dependent form of PO_4 isotherm (Barrow 1980), is given by:

$$S = kC^a\tau^b \tag{6}$$

where S and C are concentrations of solute in the sorbed and solution phase respectively, and k, a and b are empirical constants.

Using these relationships travel time (τ) of PO_4 for a vertical distance of X can be approximated (Sharma *et al.*, 1991) by:

$$\tau = \frac{X\rho}{q} kaC^{a-1}\tau^b \tag{7}$$

TABLE VII

ESTIMATED TRAVEL TIMES (τ-YRS) FOR MOVEMENT OF PHOSPHATE PATH THROUGH 1 m DEPTH OF BASSENDEAN AND SPEARWOOD SANDS WITH AN ASSUMED RECHARGE RATE (q = 100 mm/yr) AND FOR A RANGE OF C (PO_4-P CONCENTRATIONS)

C (mg/l)	Spearwood Sand τ (yrs)	Bassendean Sand
2	255	20
5	120	10
10	68	5
50	18	1
100	10	< 1

TABLE VIII
DEPTH DISTRIBUTION OF TOTAL PO_4-P (0.1 n H_2SO_4 EXTRACTABLE) IN THE SOIL PROFILES AT THE FOUR EXPERIMENTAL SITES.

Site	Depth cm	PO_4-P fraction per oven dry soil µg/g	PO_4-P Storage kg/ha	Cumulative PO_4-P Storage kg/ha
Tuart Hill	0 - 20	19.4	65.9	65.9
	20 - 40	1.6	5.6	71.4
	40 - 60	3.1	10.6	82.1
	60 - 80	5.9	20.0	102.1
	80 - 100	5.2	17.7	119.8
	100 - 120	4.0	13.5	133.4
	120 - 140	2.5	8.5	141.8
	140 - 160	1.8	6.0	147.8
	160 - 180	0.8	2.6	150.4
	180 - 200	3.2	10.9	161.3
	200 - 220	0.8	2.7	164.0
Noranda	0 - 20	45.6	155.2	155.2
	20 - 40	5.4	18.5	173.7
	40 - 60	4.0	13.7	187.4
	60 - 80	0.4	1.5	188.8
	80 - 100	0.7	2.4	191.2
	100 - 120	0.4	1.2	192.4
	120 - 140	0.3	1.1	193.6
	140 - 160	0.3	1.1	194.6
	160 - 180	0.3	0.9	195.5
	180 - 200	0.3	1.0	196.5
	200 - 220	1.6	5.4	202.0
Karrinyup	0 - 20	49.9	169.6	169.6
	20 - 40	22.4	76.2	245.8
	40 - 60	9.1	30.9	276.7
	60 - 80	4.4	15.0	291.7
	80 - 100	1.5	5.3	297.0
	100 - 120	0.9	3.1	300.1
	120 - 150	1.1	3.7	303.8
Mount Lawley	0 - 20	21.2	72.0	72.0
	20 - 40	6.2	21.2	93.3
	40 - 60	1.7	5.9	99.2
	60 - 80	2.3	8.0	107.2
	80 - 100	2.2	7.4	114.6
	100 - 120	3.0	10.0	124.6
	120 - 150	3.1	10.5	135.1

Travel times for Bassendean and Spearwood sands were computed using phosphate isotherm parameters (Barrow 1980; Sharma *et al.*, (unpublished data)). As can be appreciated from Eqn. (7) travel times are highly dependent on recharge and C. At a given recharge (q = 100 mm/yr) times for PO_4 movement to a distance fo 1 m for the two soils are compared at a number of PO_4 concentrations (Table VII).

It is quite clear that time required for PO_4 peak to travel through a Spearwood sand is much longer than through a Bassendean sand. It is expected that considerably higher times will be needed to leach the adsorbed phosphate from soil by the desorption process (Gerritse and Schofield 1989).

The measured phosphate profiles show that phosphate peak has not passed the root zone for most sites (Table VIII).

5. SUMMARY AND CONCLUSIONS

Four instrumented urban sites have been used to study the leaching of water and nutrients beneath typically managed lawns. Two domestic and two public sites were included in the study. Two suction-drained lysimeters were employed at each site.

Three of the sites are watered from bores. As many as 25% of all domestic lawns are watered in this way as are the majority of the public parks.

Over a measurement period of 210 days, irrigation amounted to between 50 and 65% of the total input, while 30 to 70% of this water passed below the root zone, carrying solutes with it. During the irrigation-dominated summer period, irrigation accounted for 55 to 75% of the total water input, of which 15 to 70% passed below the root zone. During summer, total water input (irrigation + precipitation) was between 82 and 111% of evaporation from Class A Pan.

Solute balances were performed for chloride, nitrate-N, phosphate-P and ammonium-N. Recharge values estimated based on chloride mass balance were slightly lower than those measured by lysimeters. Chloride input originated from irrigation, precipitation and from fertilisers. Better estimates of the chloride input in irrigation and precipitation and better accounting for changes in soil chloride storage should provide better estimate of recharge.

In the case of the two domestic lawns, for the same fertiliser inputs, flow-weighted total nitrogen (NO_3-N + NH_4-N) concentrations in leachate were 1.98 mg/l and 10.11 mg/l , while flow-weighted PO_4-P concentrations were 0.026 mg/l and 0.022 mg/l. Ammonium accounted for only about 10% of the total inorganic N leached.

In the case of the two public sites, flow-weighted total nitrogen concentrations were 4.56 mg/l and 10.44 mg/l, with ammonium accounting for < 10% of the total inorganic N leached. Flow-weighted PO_4-P concentrations were 0.125 mg/l and 0.135 mg/l.

In general, a marked seasonal variation was exhibited in solute concentrations in leachate. On occasions, beneath most urban lawn sites, leachate contained NO_3-N concentrations exceeding the recommended WHO limit for drinking water of 10 mg/l. However, groundwater concentrations are expected to be lower due to dilution and denitrification effects. These aspects however require further research.

A mechanistic modelling approach was applied to simulate Cl, NH_4 and NO_3 transport through the unsaturated zone. A quasi-steady state model, involving a numerical solution of the dispersive-convective transport equation, was used. The dispersion coefficient, as a function of pore water velocity, was calculated through optimisation using measured chloride concentrations in the leachate. Calibration of the model (for NH_4 and NO_3) was performed through the incorporated multi-dimensional, parameter estimation technique. Model simulations of NH_4 and NO_3 concentrations at lysimeter level compared fairly well with the measured values.

A model involving a combination of solute transport equation (without the dispersion term) and time dependent phosphate isotherm was applied to estimate travel time for phosphate through soil profile. Using this model, it was estimated that for a soil supplied with a phosphate solution of 5 mg/l and average recharge of 100 mm/yr, the time required for the phosphate peak to travel a distance of 1 m will be about 120 yrs through Spearwood and about 10 yrs through Bassendean sand. These computations were supported by field observations. Our experimental data show that some PO_4 appears to leak through the system under heavy water inputs.

6. RECOMMENDATIONS

Our results clearly show that with the current management practices used in the urban Perth area, there is substantial transport of nitrogen and phosphorus below the root zone. The leached nutrients are a threat to many urban wetlands, and especially to the Swan-Canning river system. Observations show that over-irrigation, and over-fertilisation are being practiced in managing public as well as private urban lawns. In order to reduce nutrient pollution risks, modified management practices need to be developed. Alternatively, lifestyles with reduced areas in lawns/gardens or gardens using native vegetation which require little fertilising and watering may need to be considered.

7. ACKNOWLEDGMENTS

This research was conducted with the financial support of the Water Authority of Western Australia and the Urban Water Research Association of Australia.

We wish to thank the owners of two domestic lawns, and the Councils managing the two public park sites, for their permission for frequent access and cooperation in providing the relevant information on these sites. We are thankful to Quentin Richards for assistance with the preparation of the report.

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9. APPENDICES

9.1 List of Figures

9.2 List of Tables

9.3 Figures

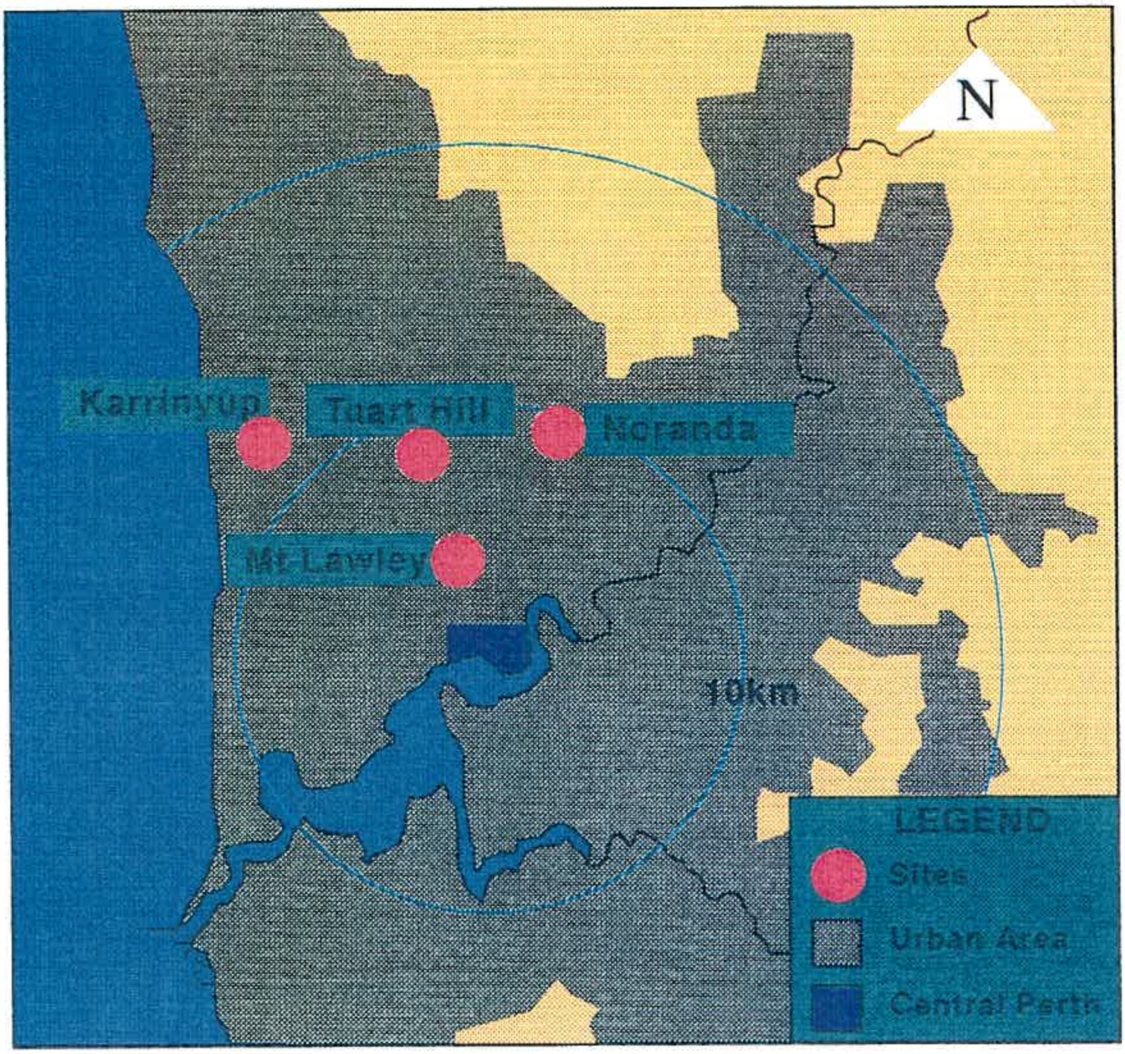

Figure 1. Location of four urban lawn sites selected for experimentation on the Swan Coastal Plain, Western Australia

LYSIMETER DESIGN

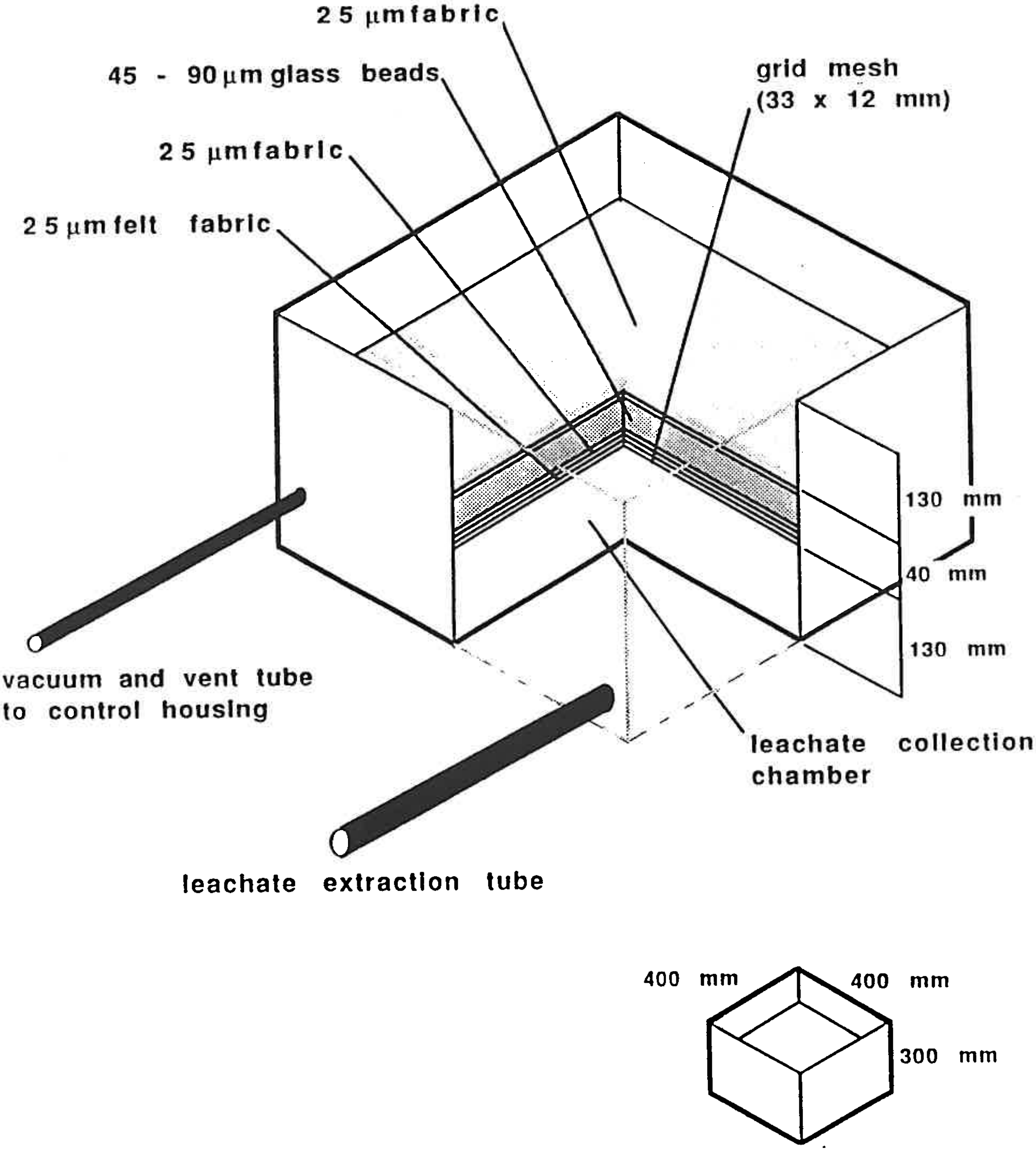

Figure 2. Design of mini-lysimeter and its associated parts, used for quantifying nutrient and water fluxes.

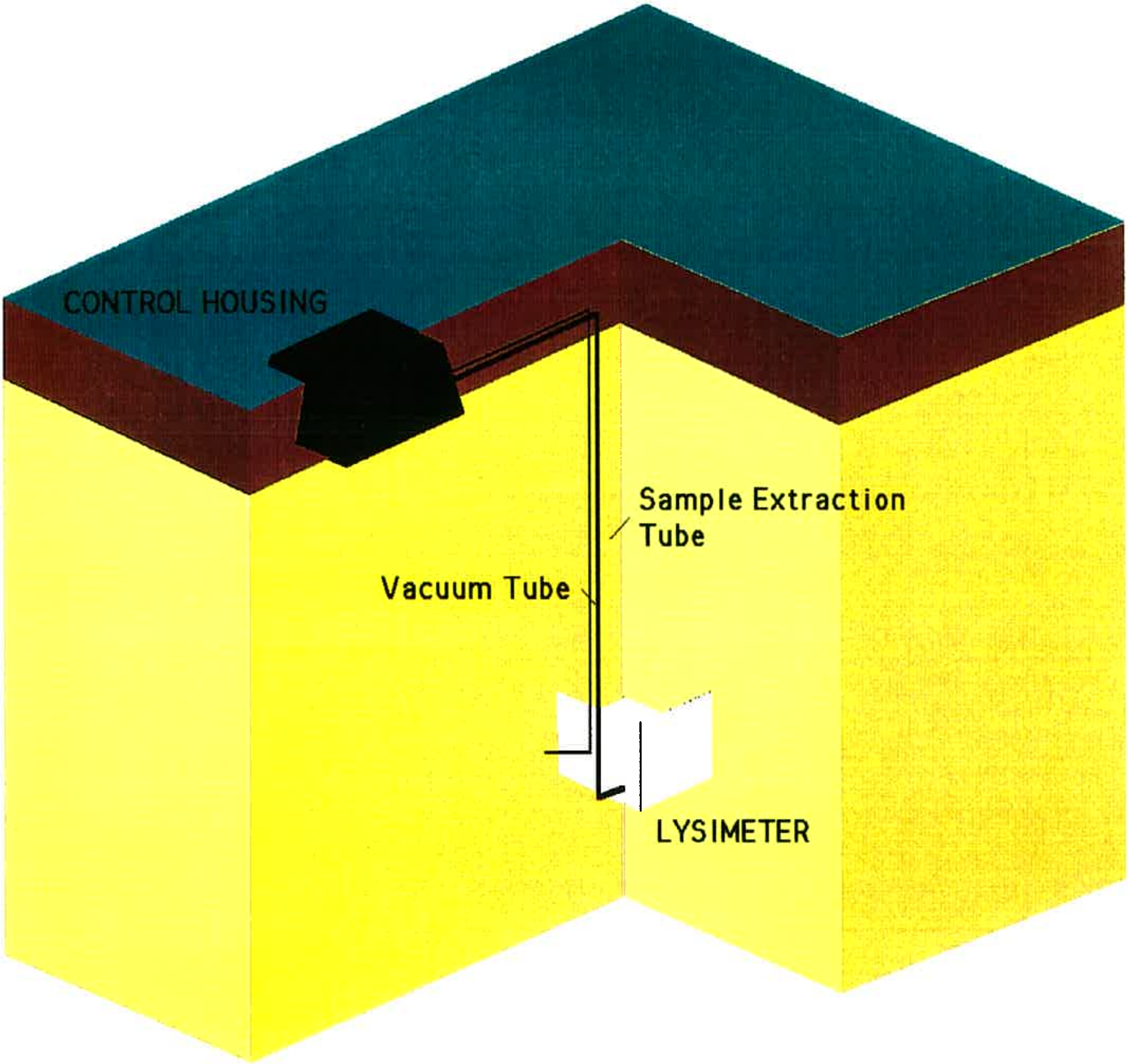

Figure 3. Installation layout of lysimeter in the soil profile.

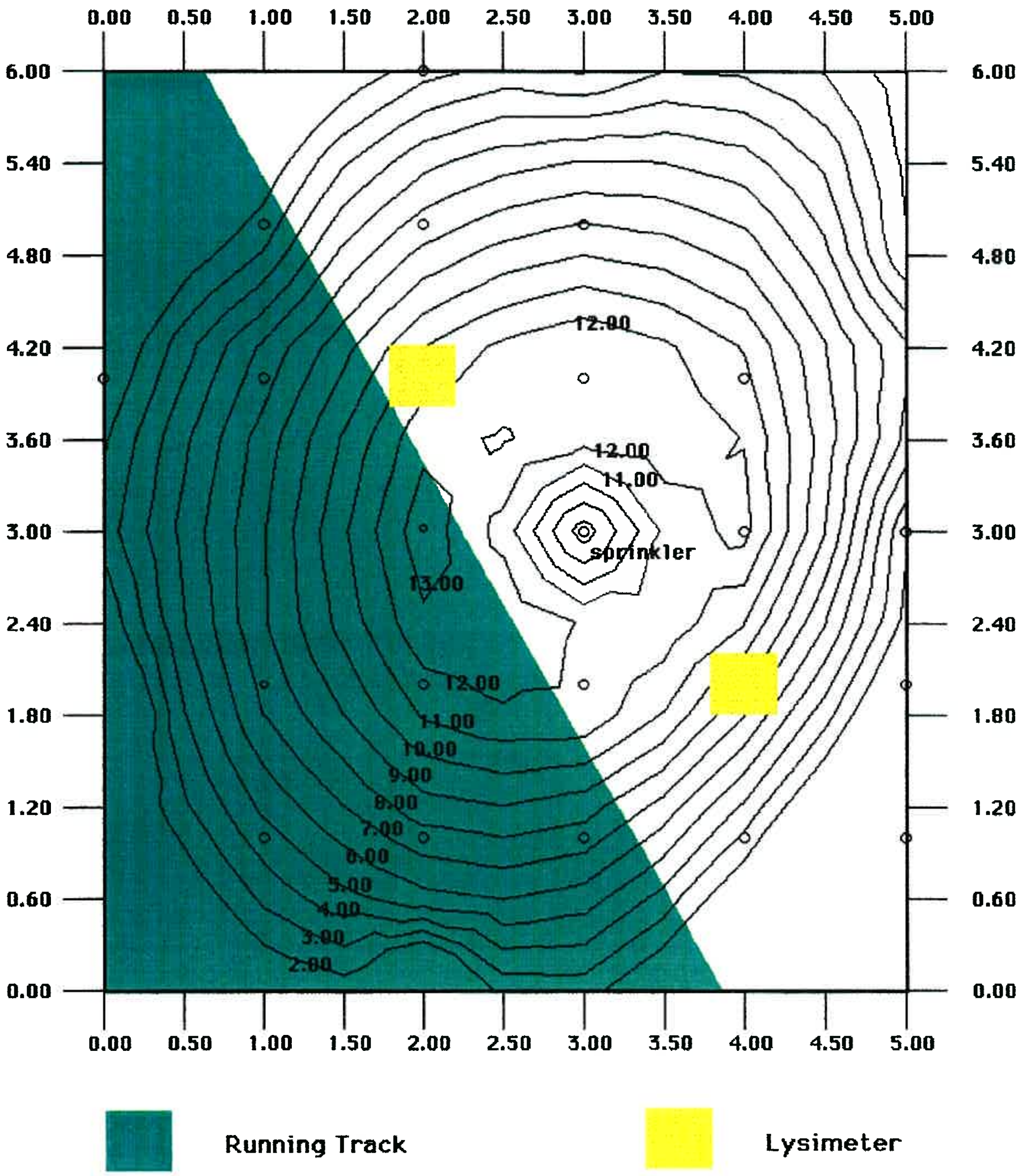

Figure 4. Areal distribution of irrigation input from a sprinkler at the Noranda public park site during an irrigation test period. The iso-irrigation contours (mm) in relation to the location of lysimeters and sprinkler are shown.

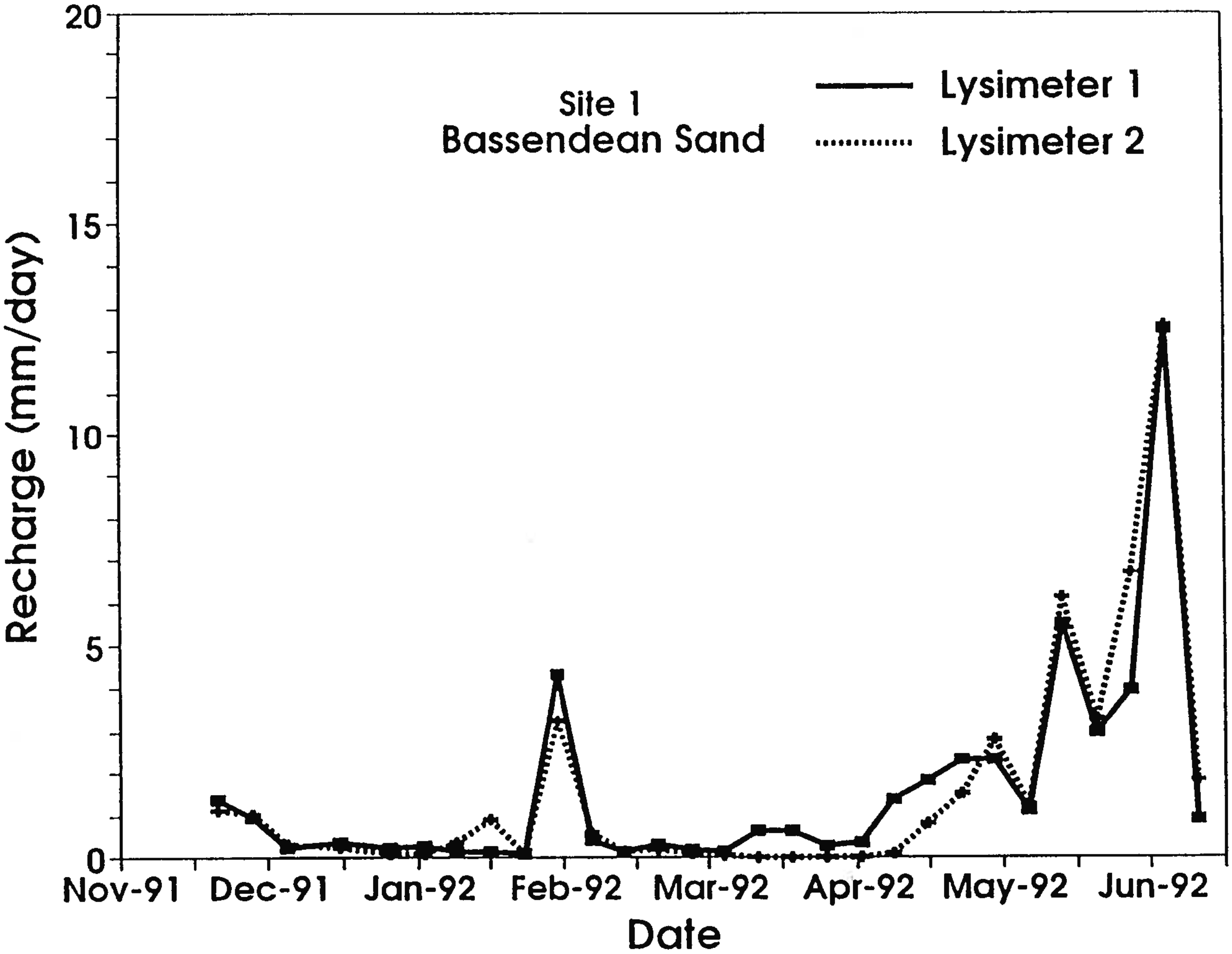

Figure 5. Recharge (water flux) measured by lysimeters at urban site 1 (Noranda).

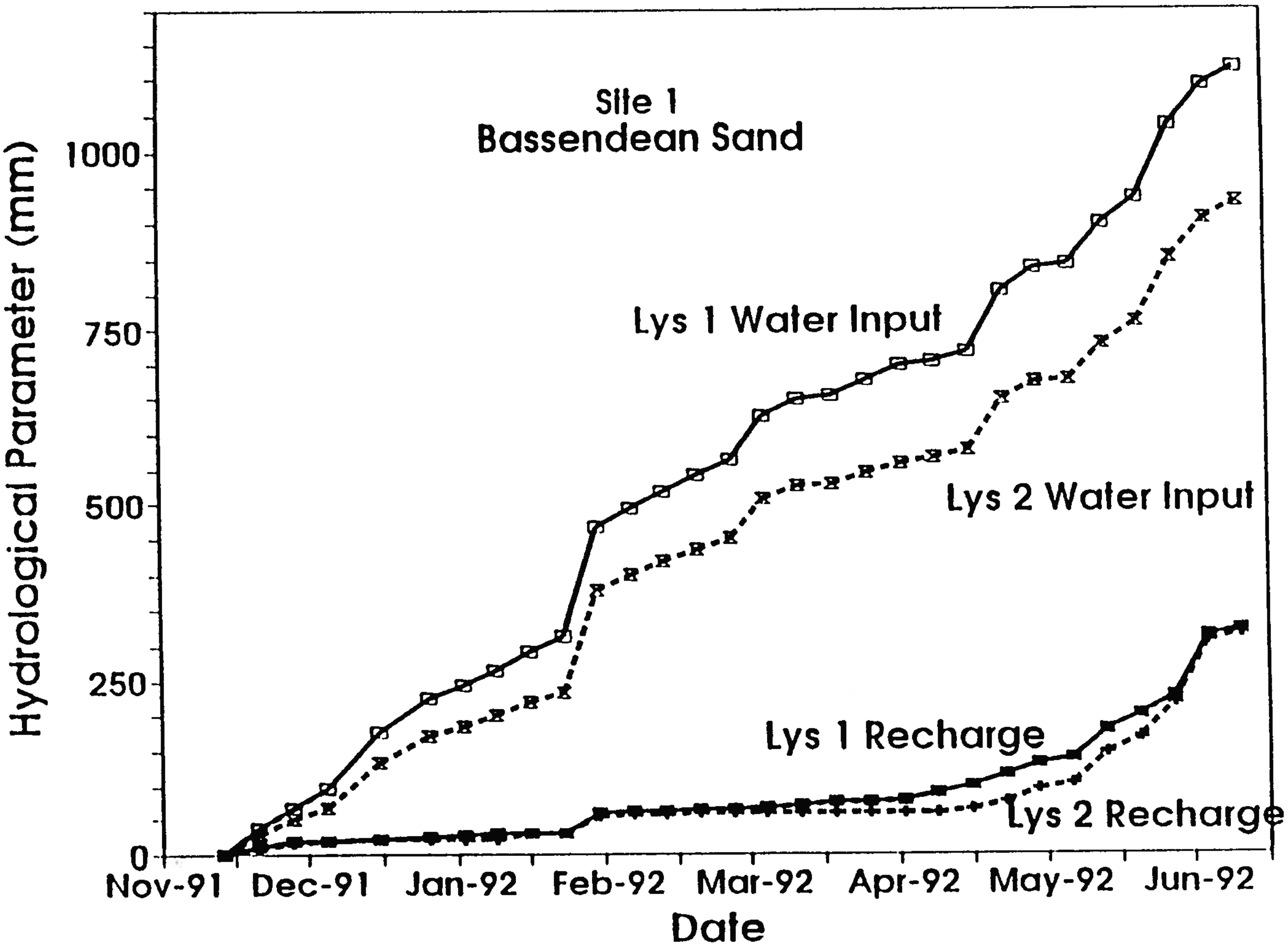

Figure 6. Cumulative recharge and total water input (irrigation + precipitation) at urban site 1 (Noranda).

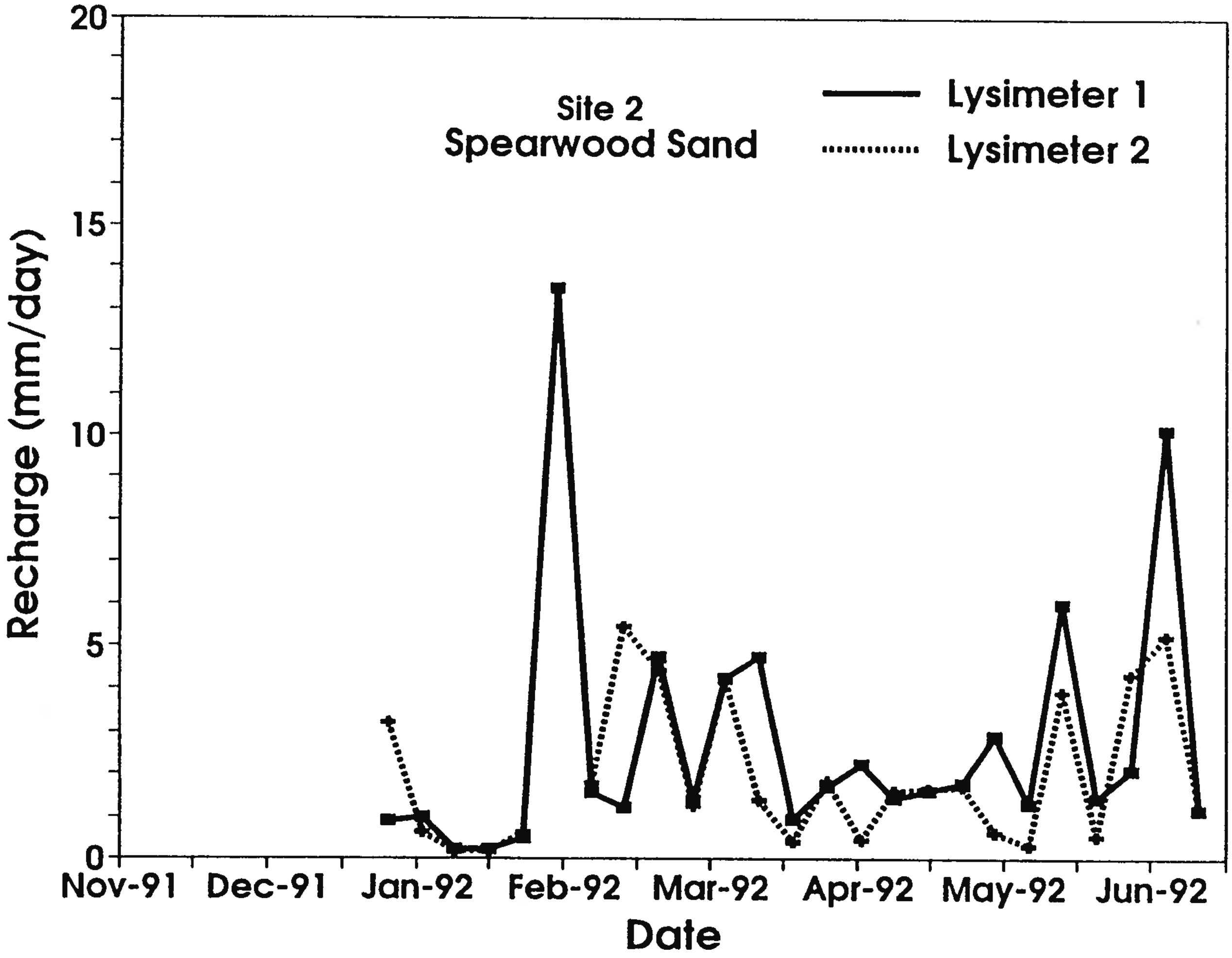

Figure 7. Recharge (water flux) measured by lysimeters at urban site 2 (Tuart Hill).

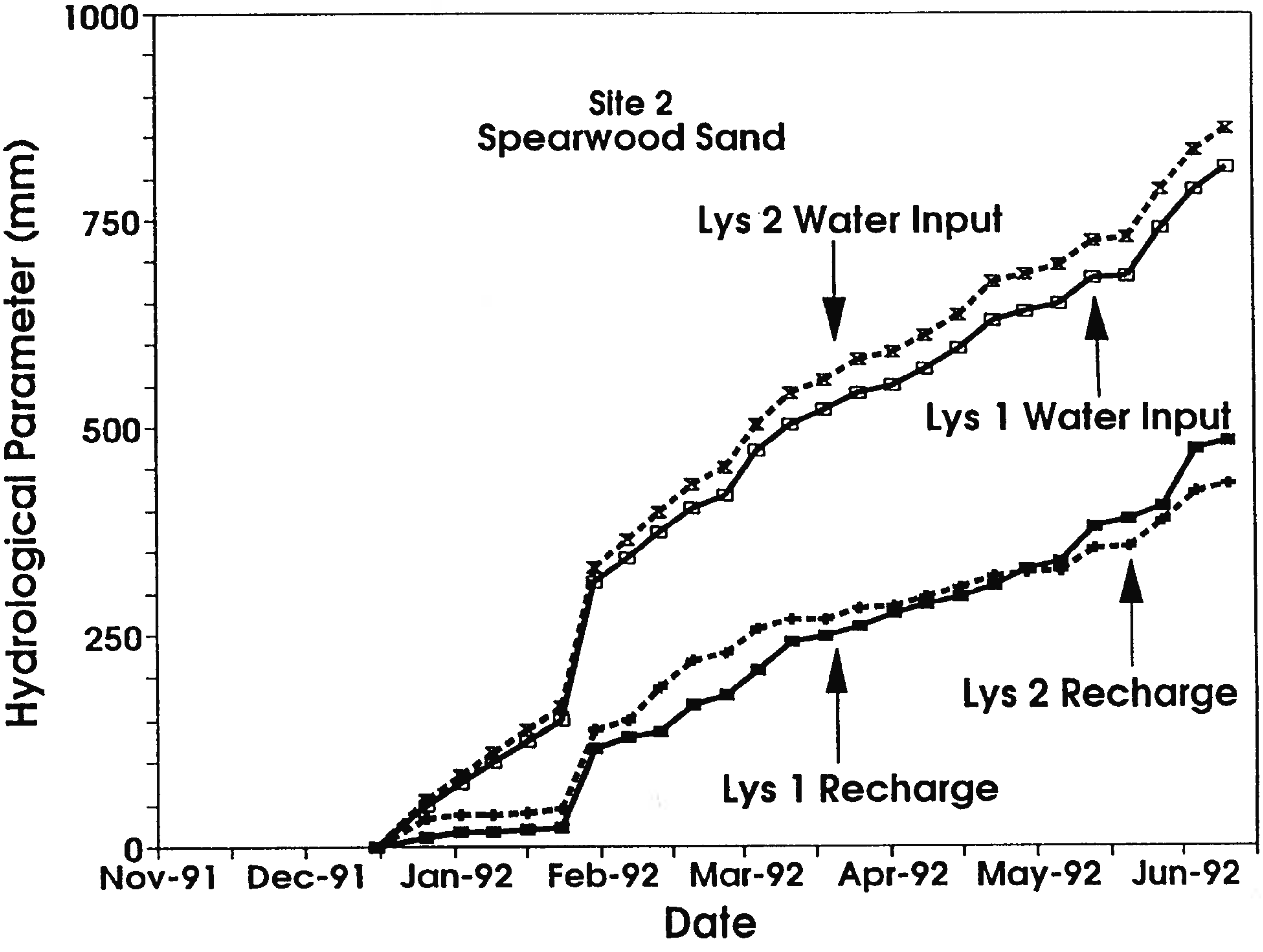

Figure 8. Cumulative recharge and total water input (irrigation + precipitation) at urban site 2 (Tuart Hill).

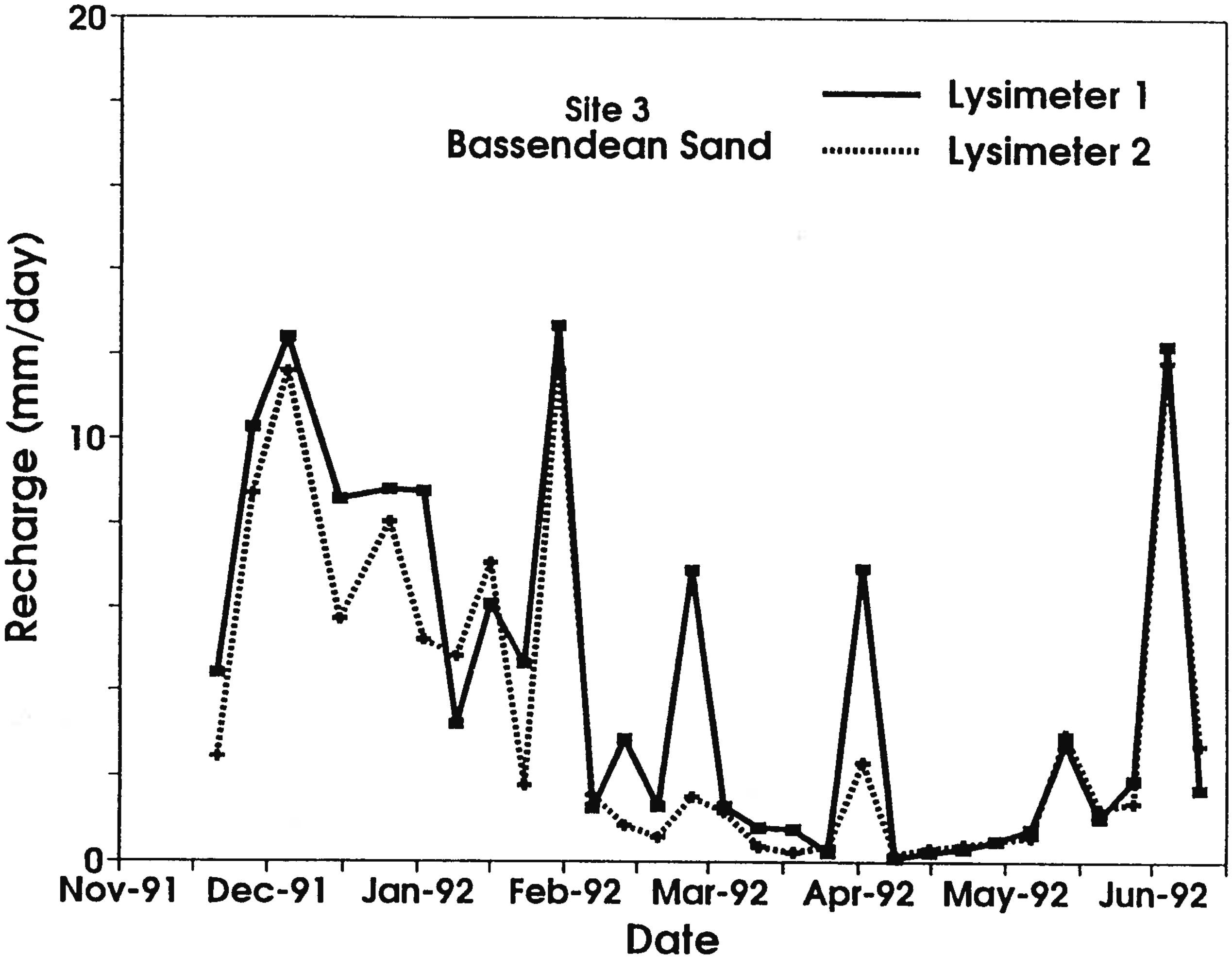

Figure 9. Recharge (water flux) measured by lysimeters at urban site 3 (Mt Lawley).

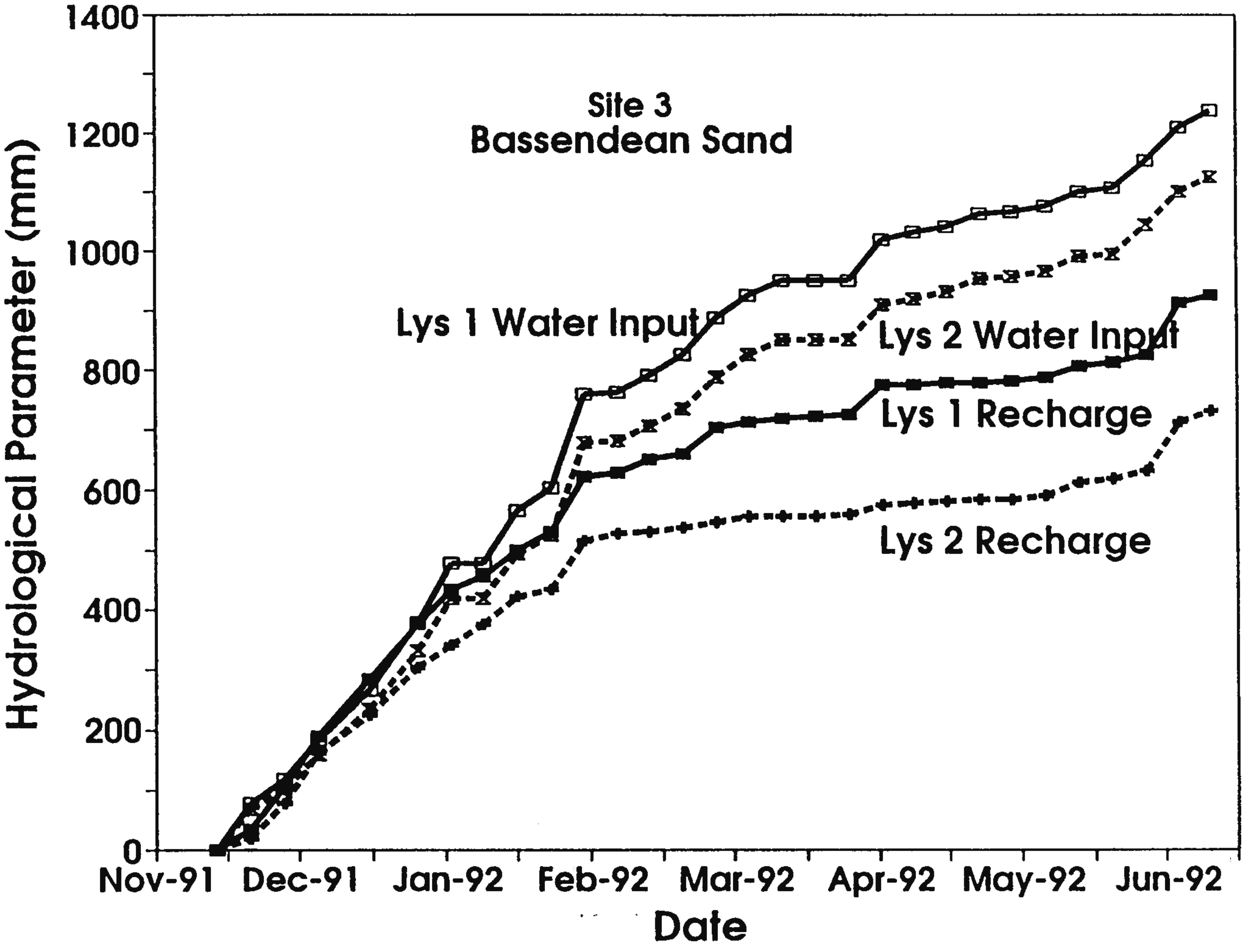

Figure 10. Cumulative recharge and total water input (irrigation + precipitation) at urban site 3 (Mt Lawley).

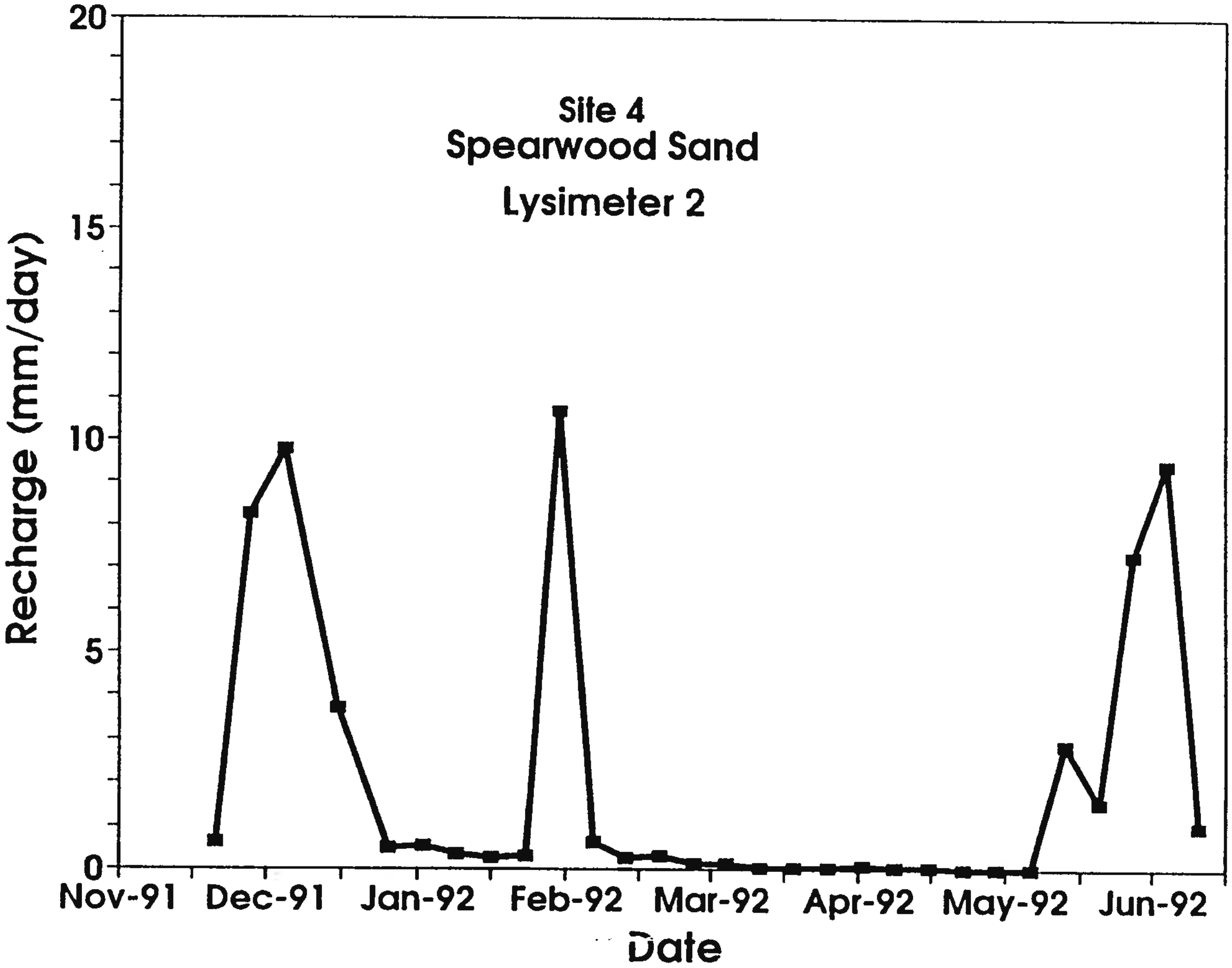

Figure 11. Recharge (water flux) measured by a lysimeter at urban site 4 (Karrinyup).

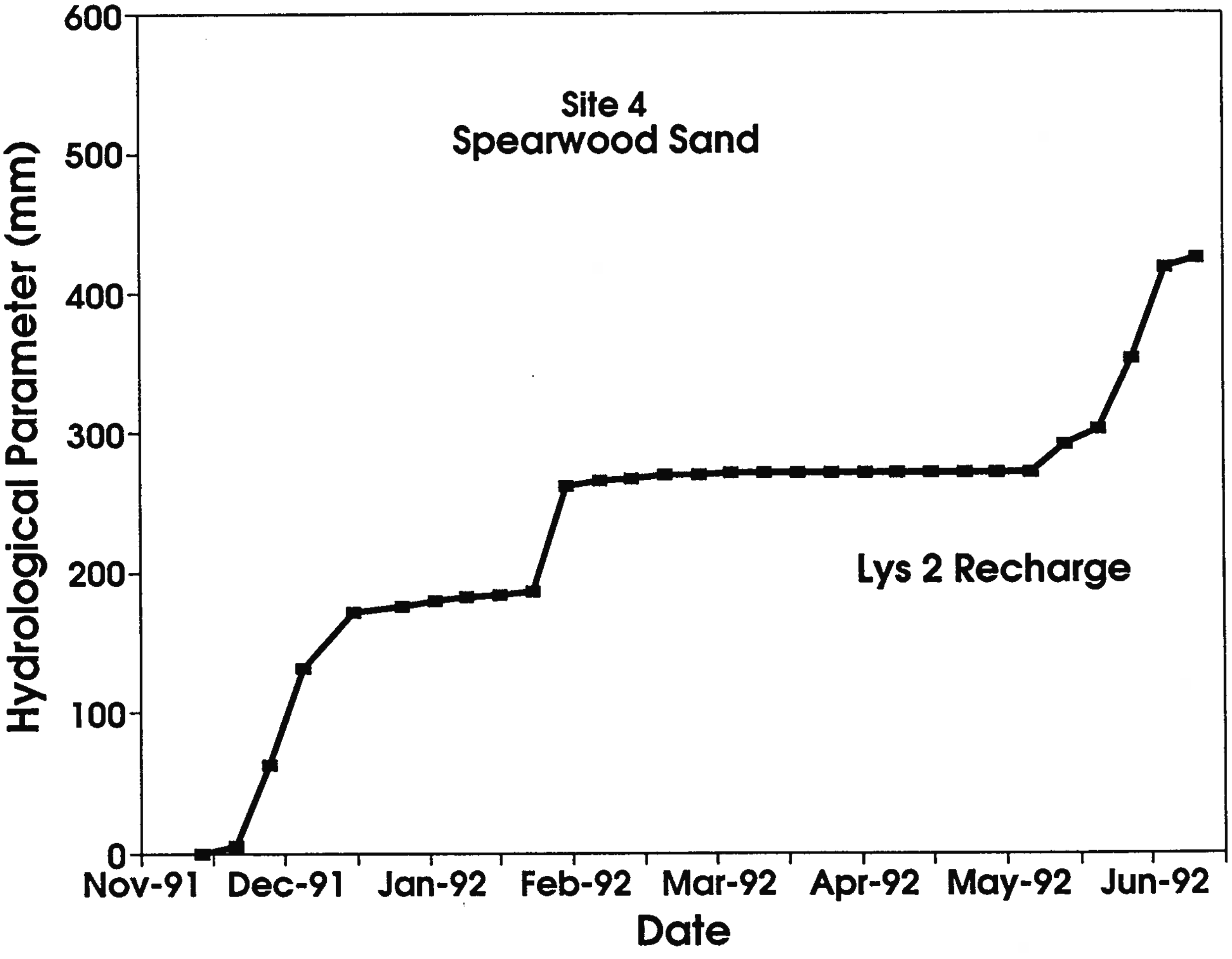

Figure 12. Cumulative recharge at urban site 4 (Karrinyup).

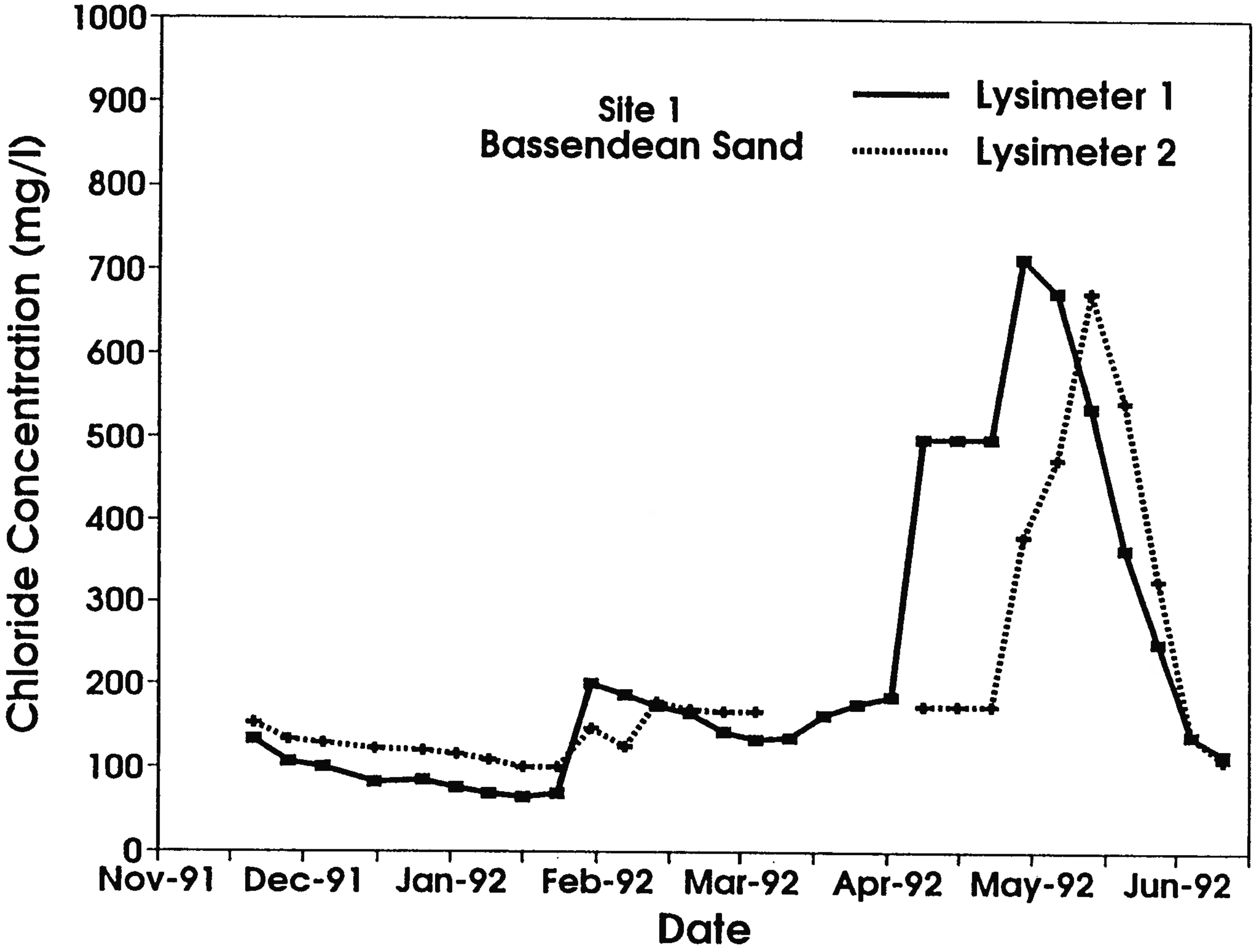

Figure 13. Chloride concentration in leachate from lysimeters at urban site 1 (Noranda).

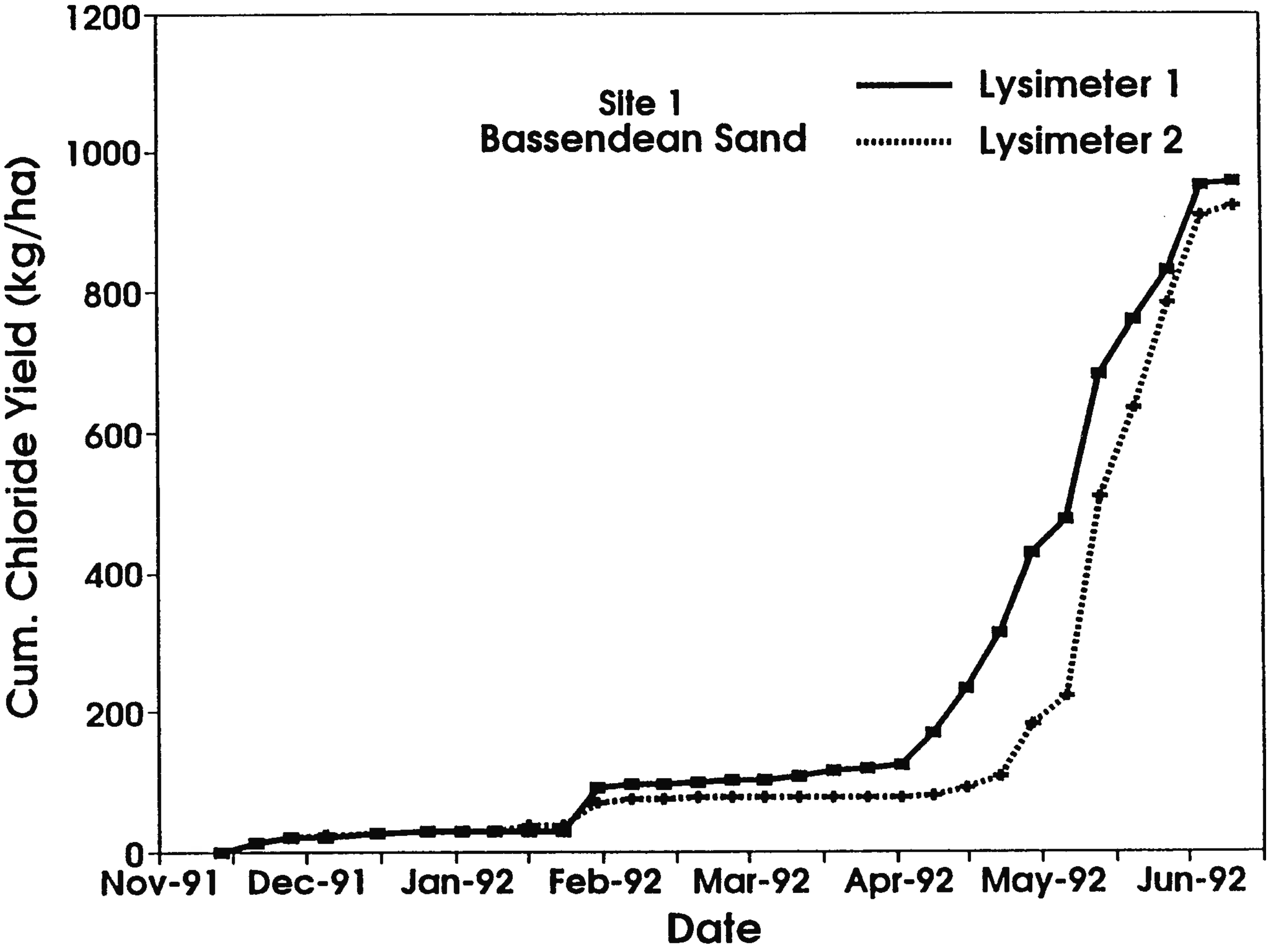

Figure 14. Chloride yield in leachate from lysimeters at urban site 1 (Noranda).

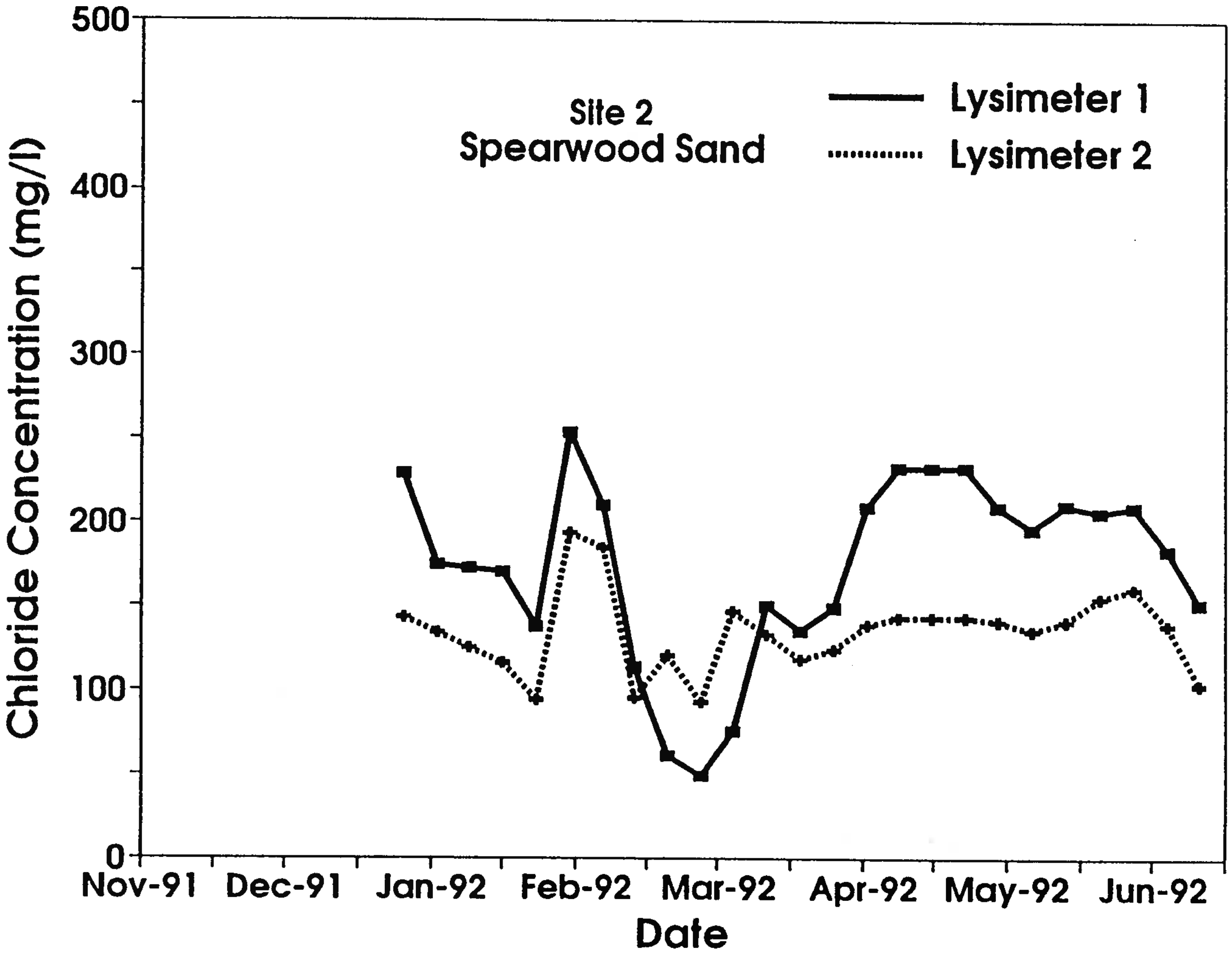

Figure 15. Chloride concentration in leachate from lysimeters at urban site 2 (Tuart Hill).

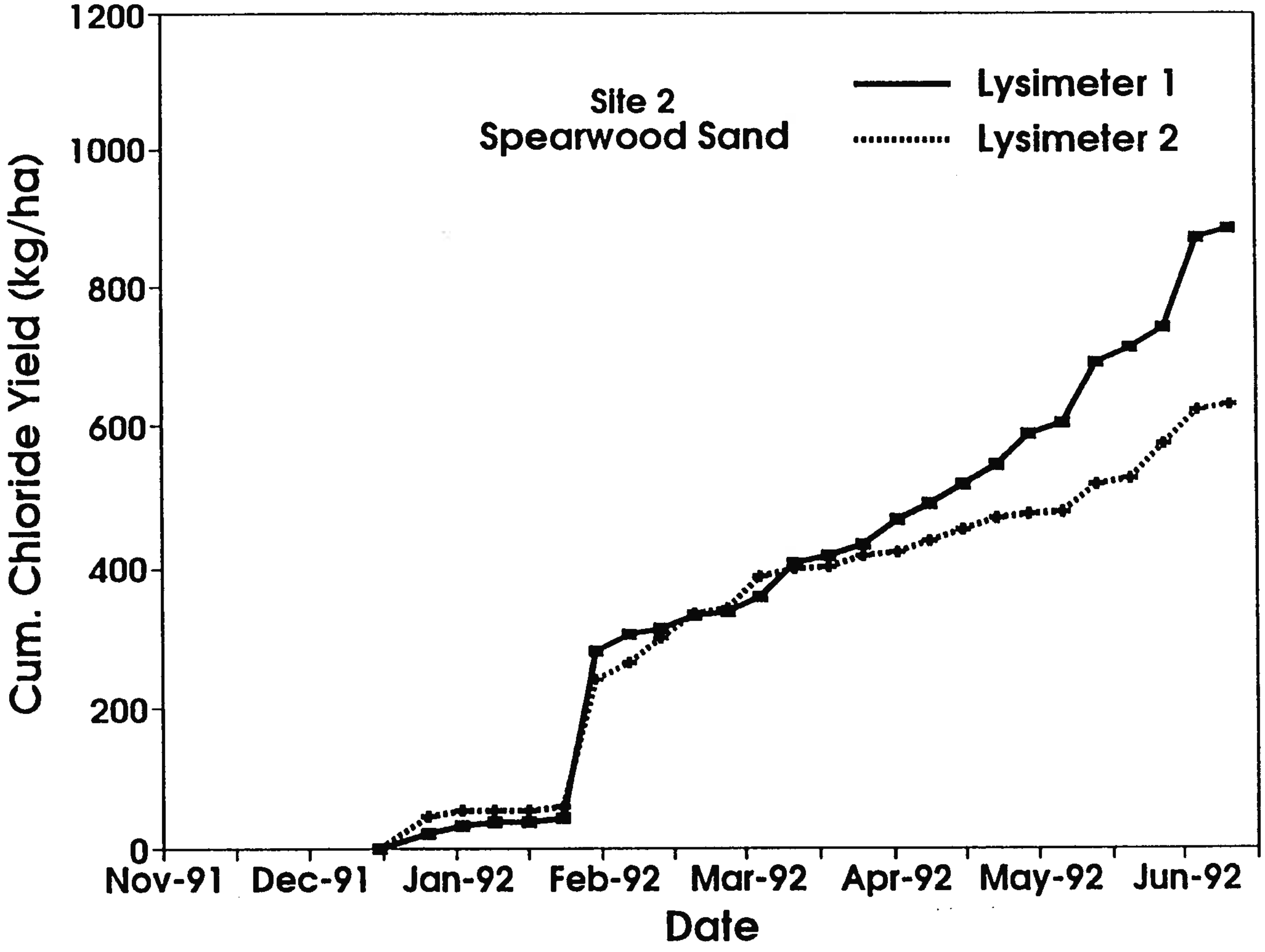

Figure 16. Chloride yield in leachate from lysimeters at urban site 2 (Tuart Hill).

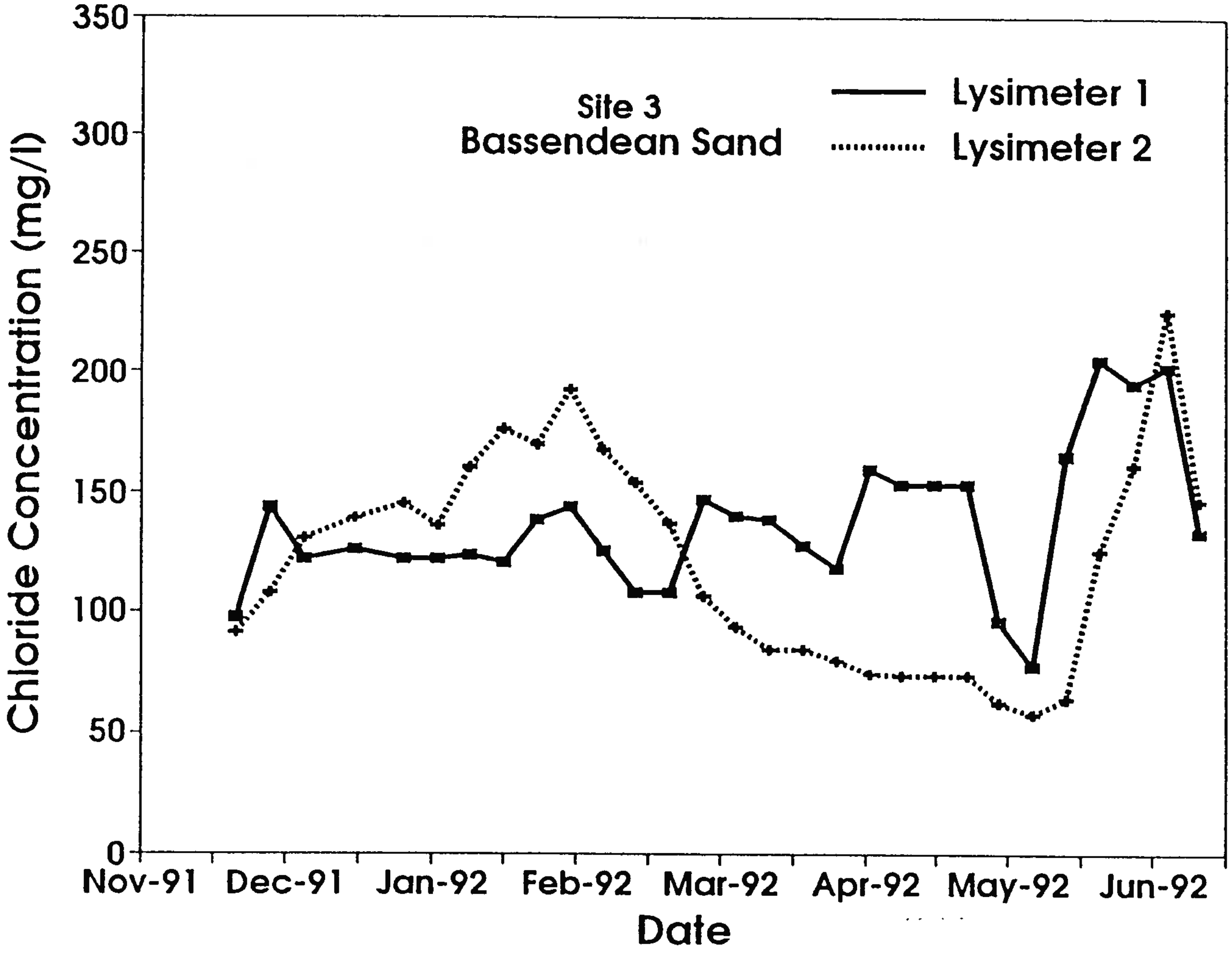

Figure 17. Chloride concentration in leachate from lysimeters at urban site 3 (Mt Lawley).

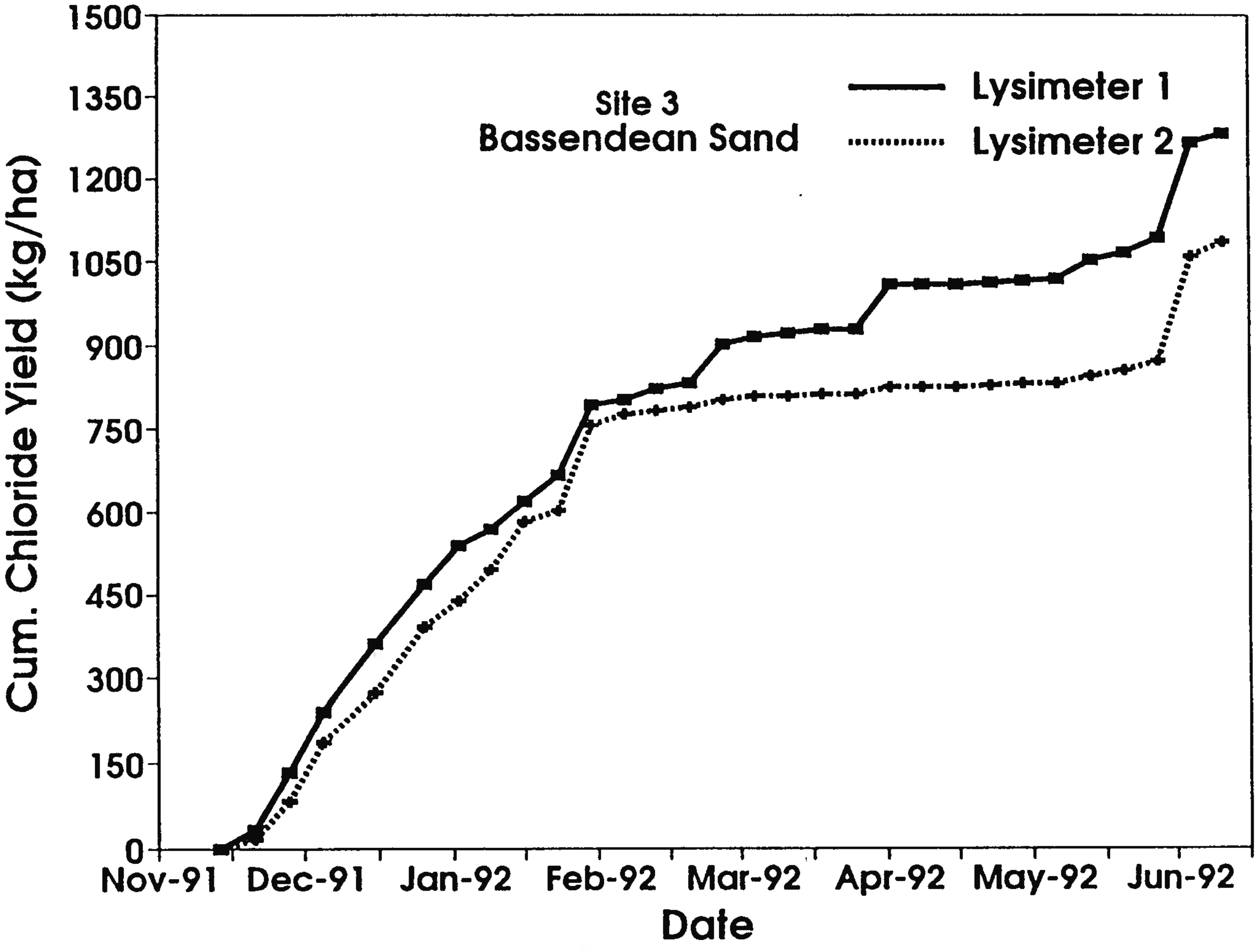

Figure 18. Chloride yield in leachate from lysimeters at urban site 3 (Mt Lawley).

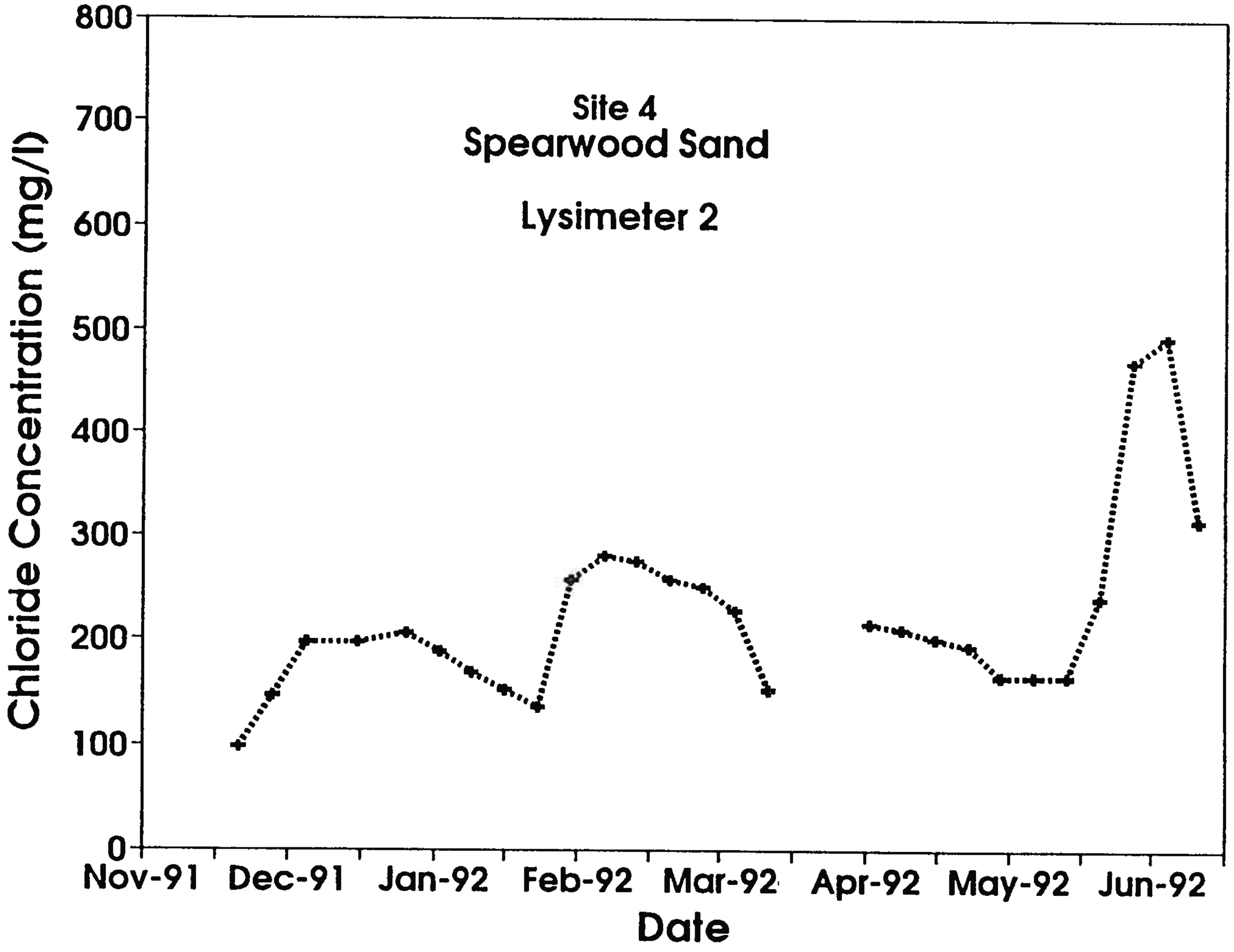

Figure 19. Chloride concentration in leachate from a lysimeter at urban site 4 (Karrinyup).

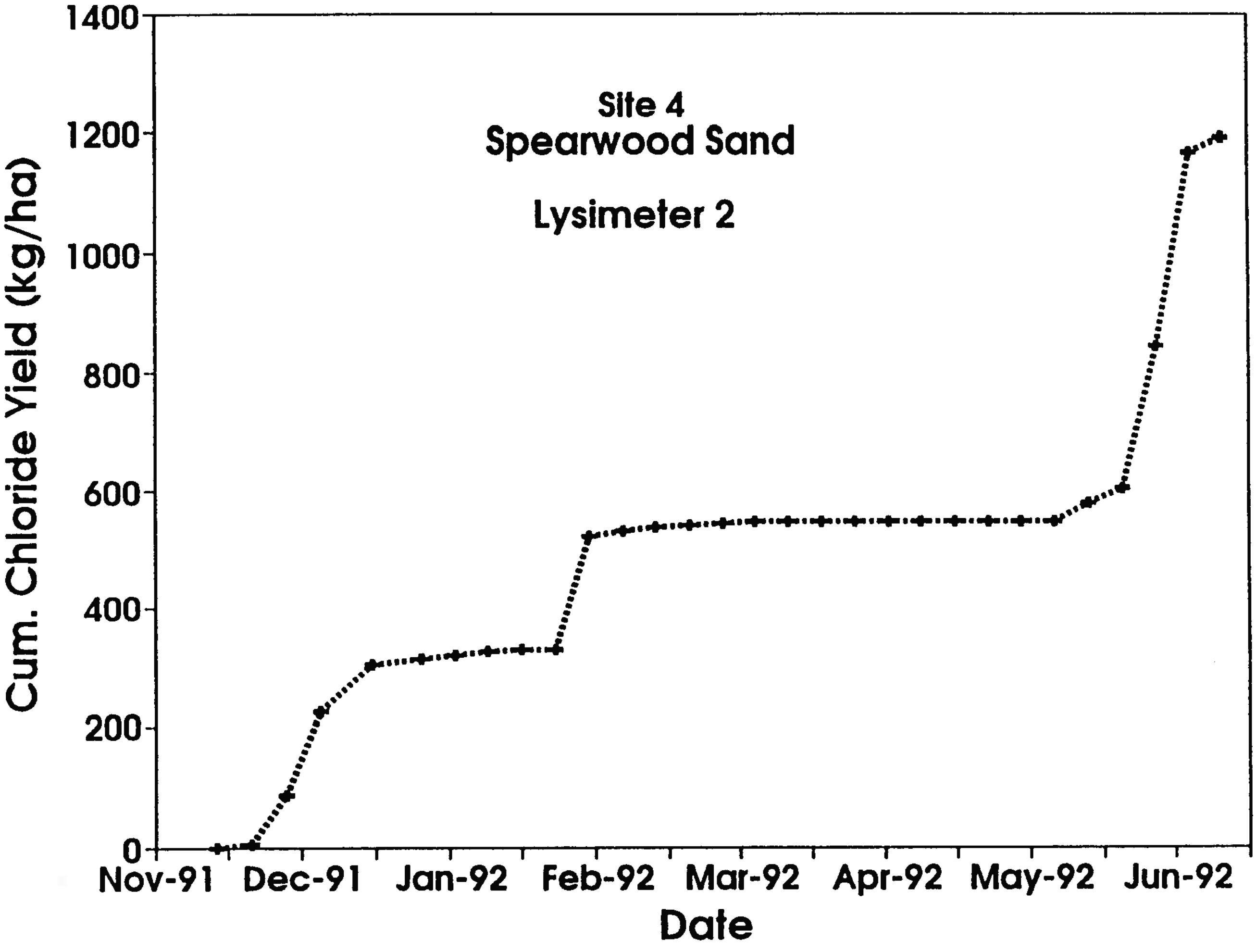

Figure 20. Chloride yield in leachate from a lysimeter at urban site 4 (Karrinyup).

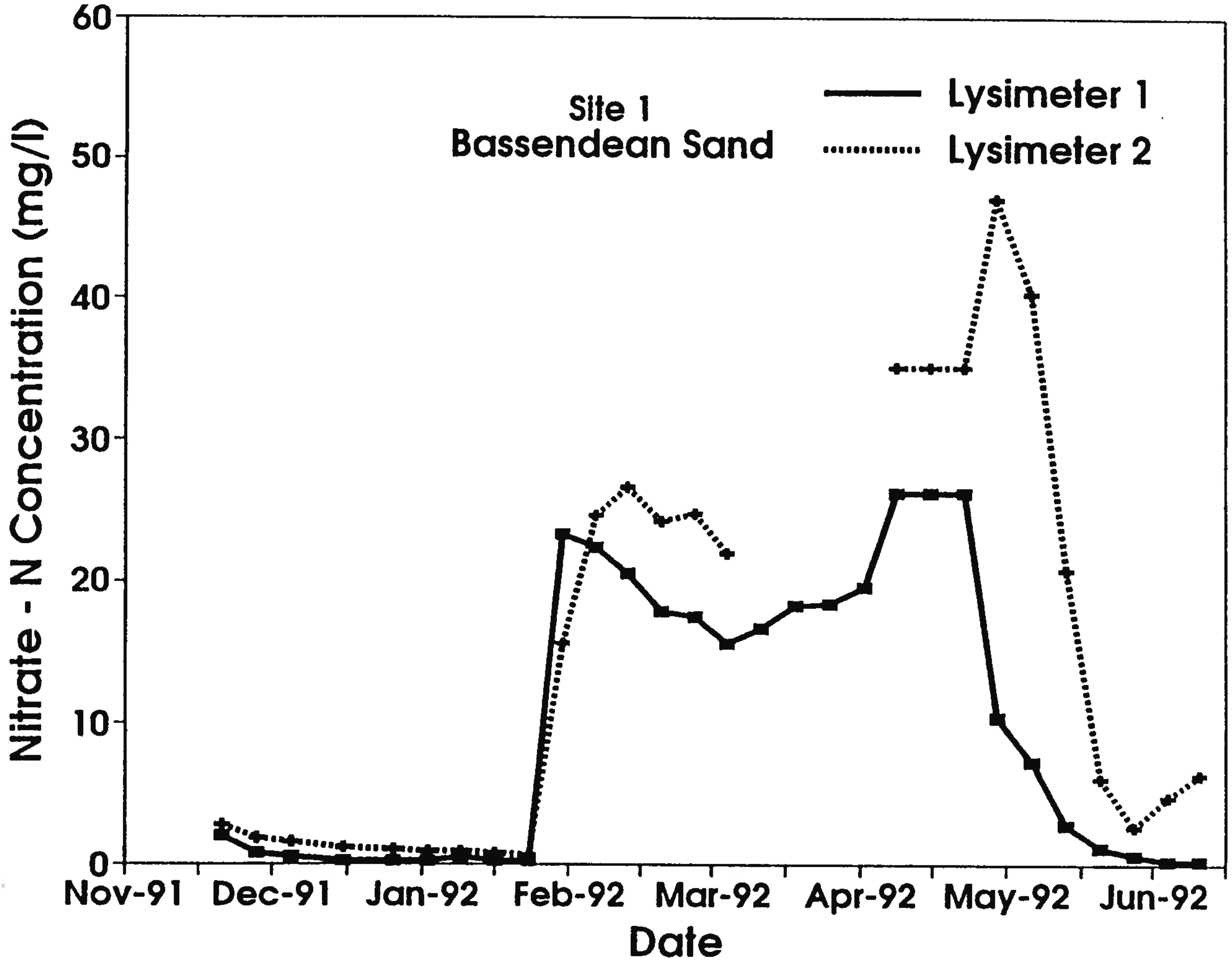

Figure 21. Nitrate-N concentration in leachate from lysimeters at urban site 1 (Noranda).

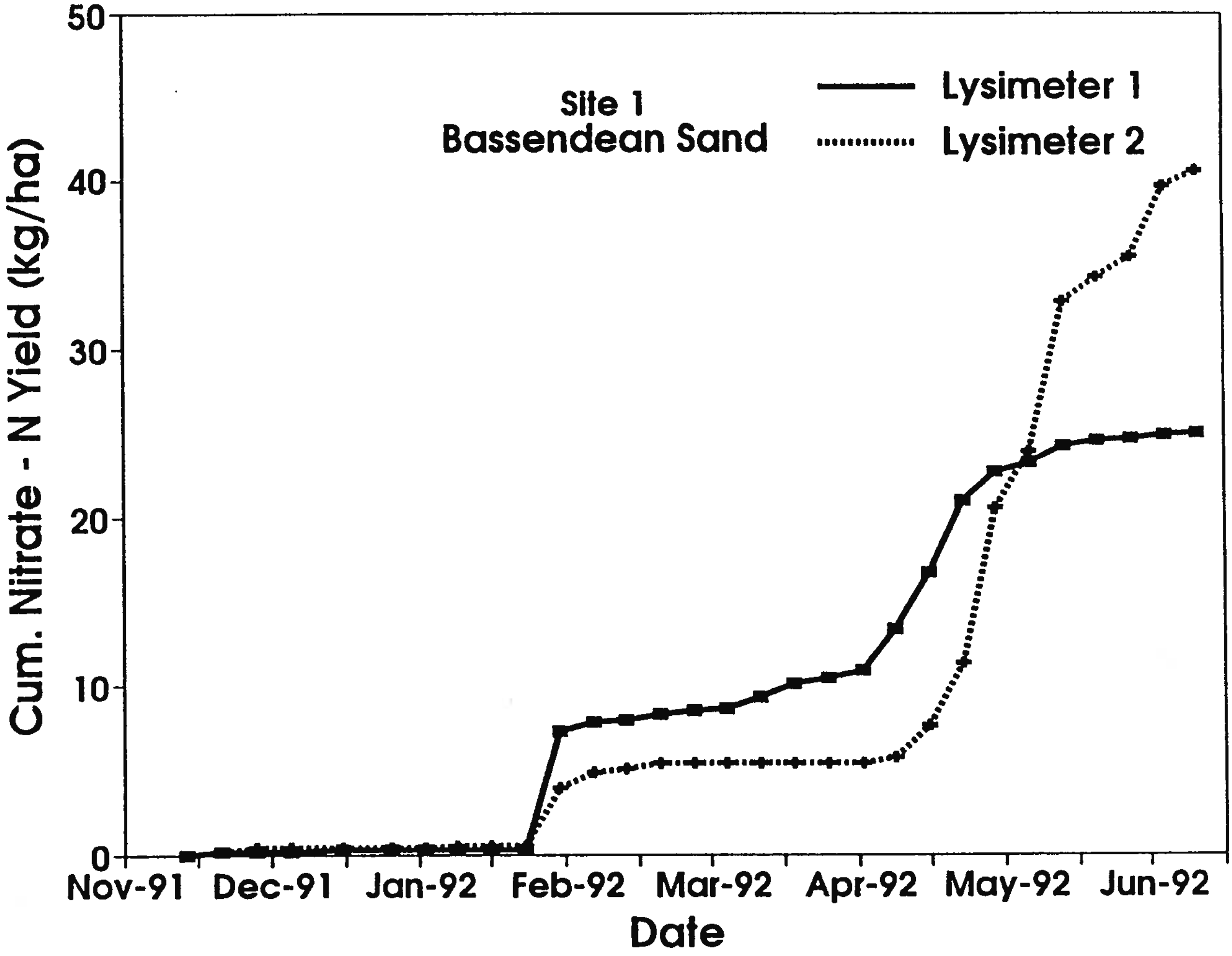

Figure 22. Nitrate-N yield in leachate from lysimeters at urban site 1 (Noranda).

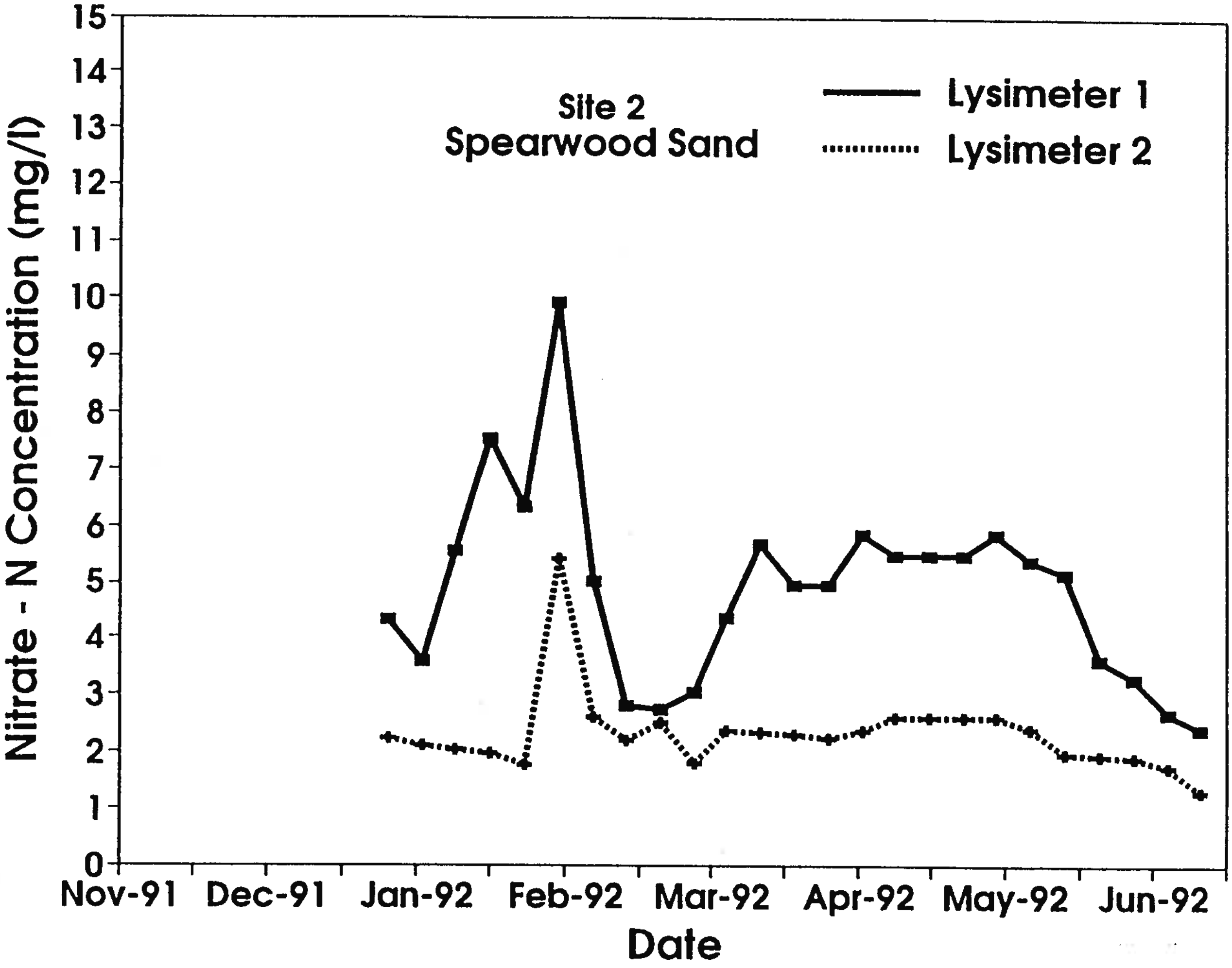

Figure 23. Nitrate-N concentration in leachate from lysimeters at urban site 2 (Tuart Hill).

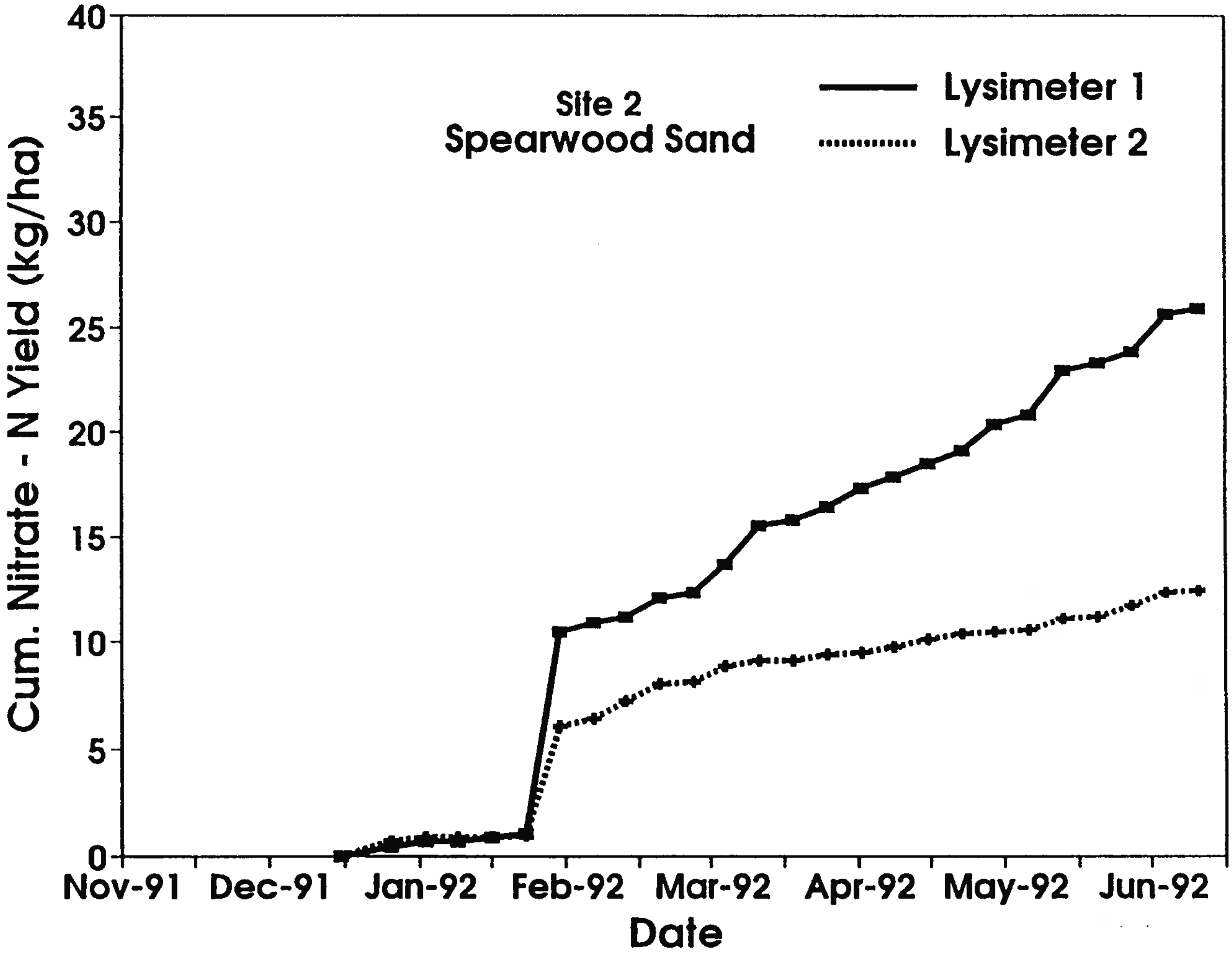

Figure 24. Nitrate-N yield in leachate from lysimeters at urban site 2 (Tuart Hill).

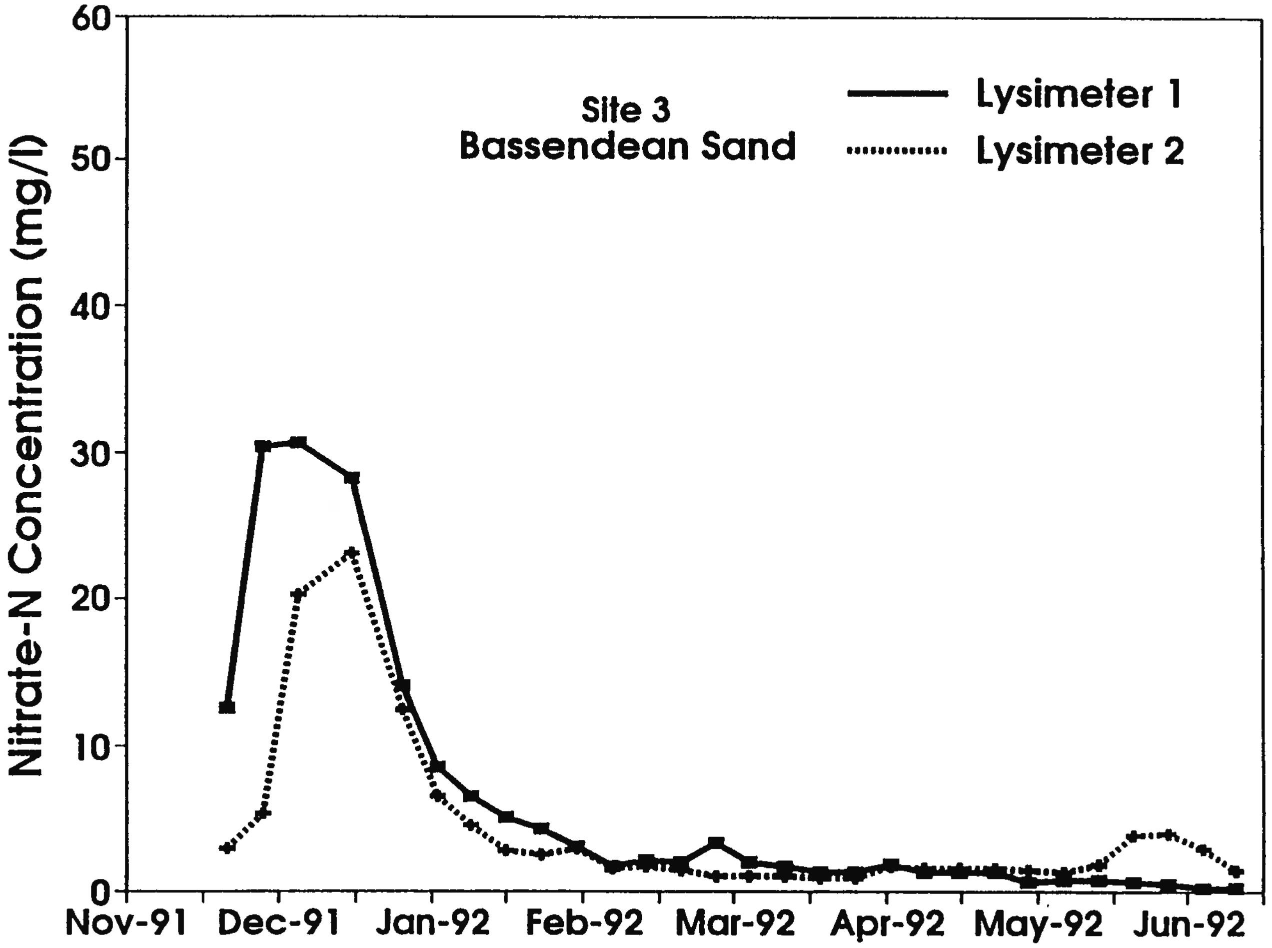

Figure 25. Nitrate-N concentration in leachate from lysimeters at urban site 3 (Mt Lawley).

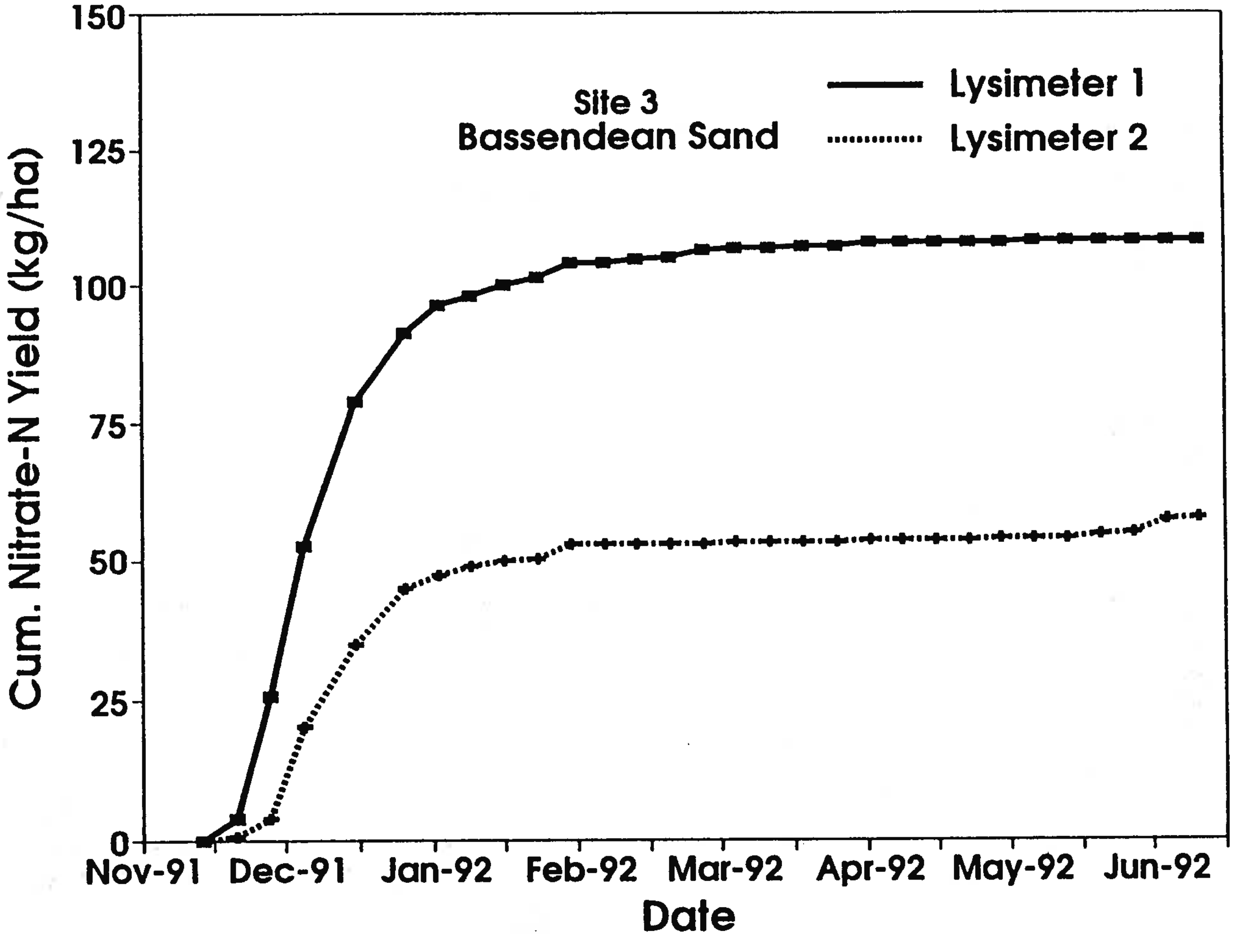

Figure 26. Nitrate-N yield in leachate from lysimeters at urban site 3 (Mt Lawley).

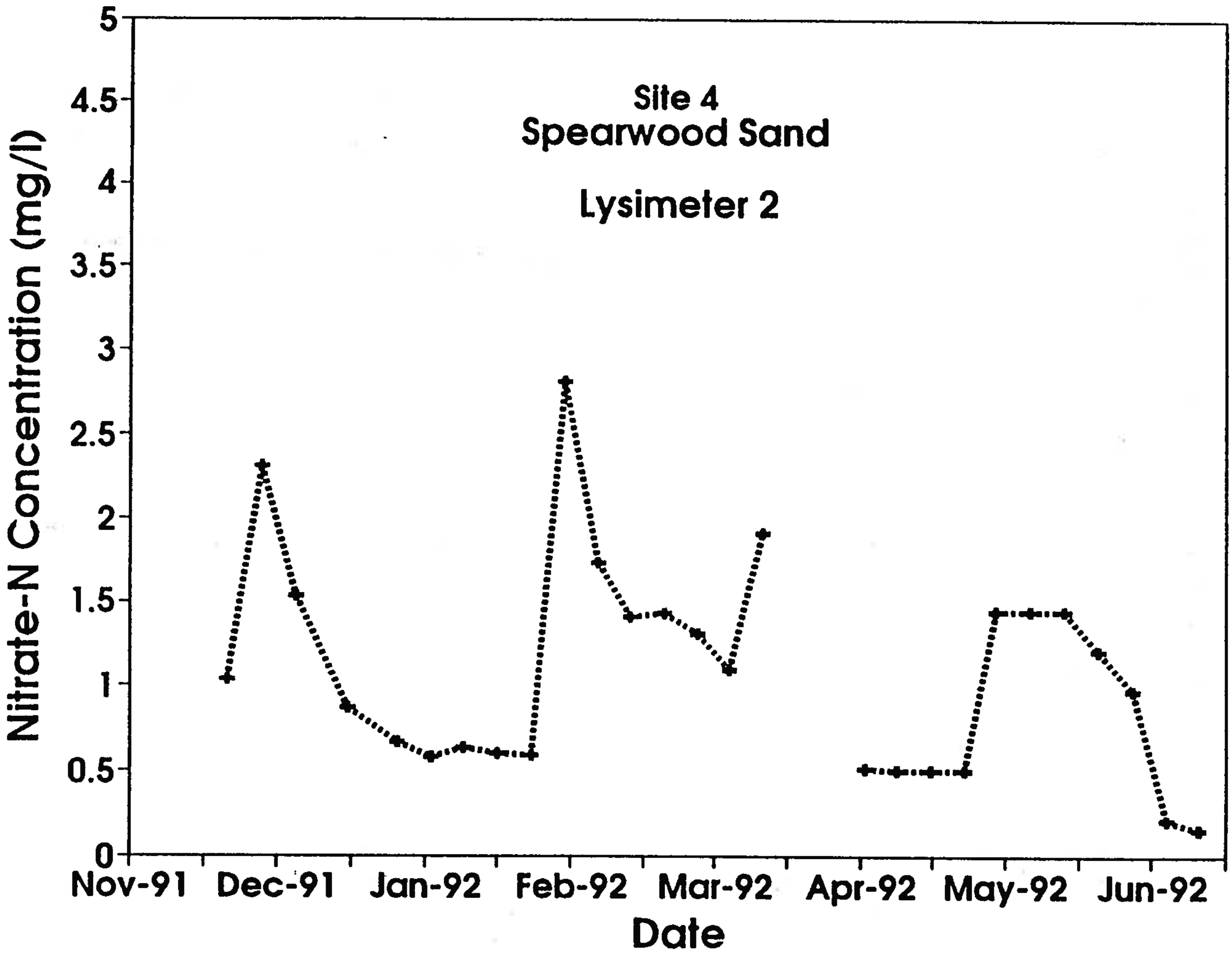

Figure 27. Nitrate-N concentration in leachate from a lysimeter at urban site 4 (Karrinyup).

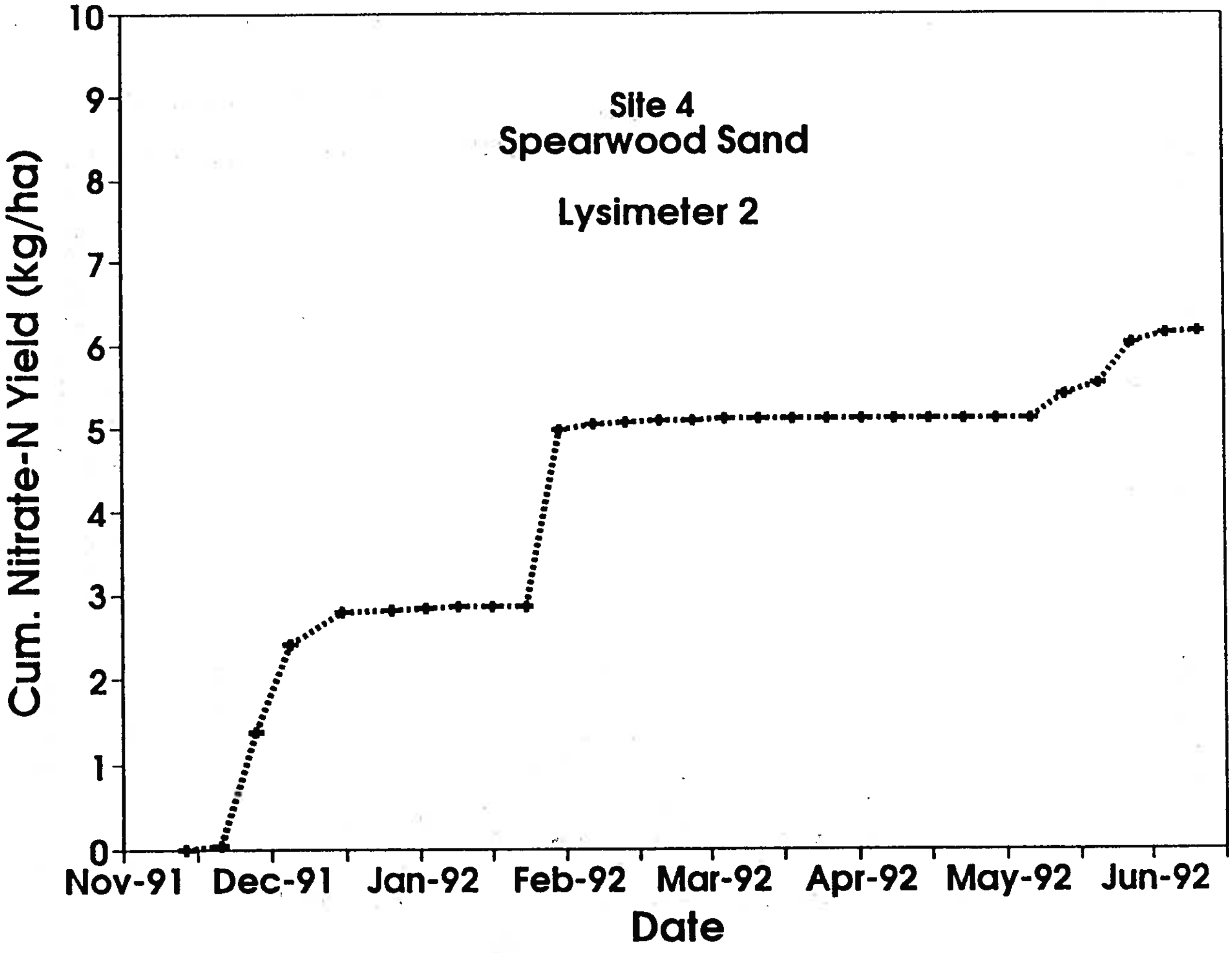

Figure 28. Nitrate-N yield in leachate from a lysimeter at urban site 4 (Karrinyup).

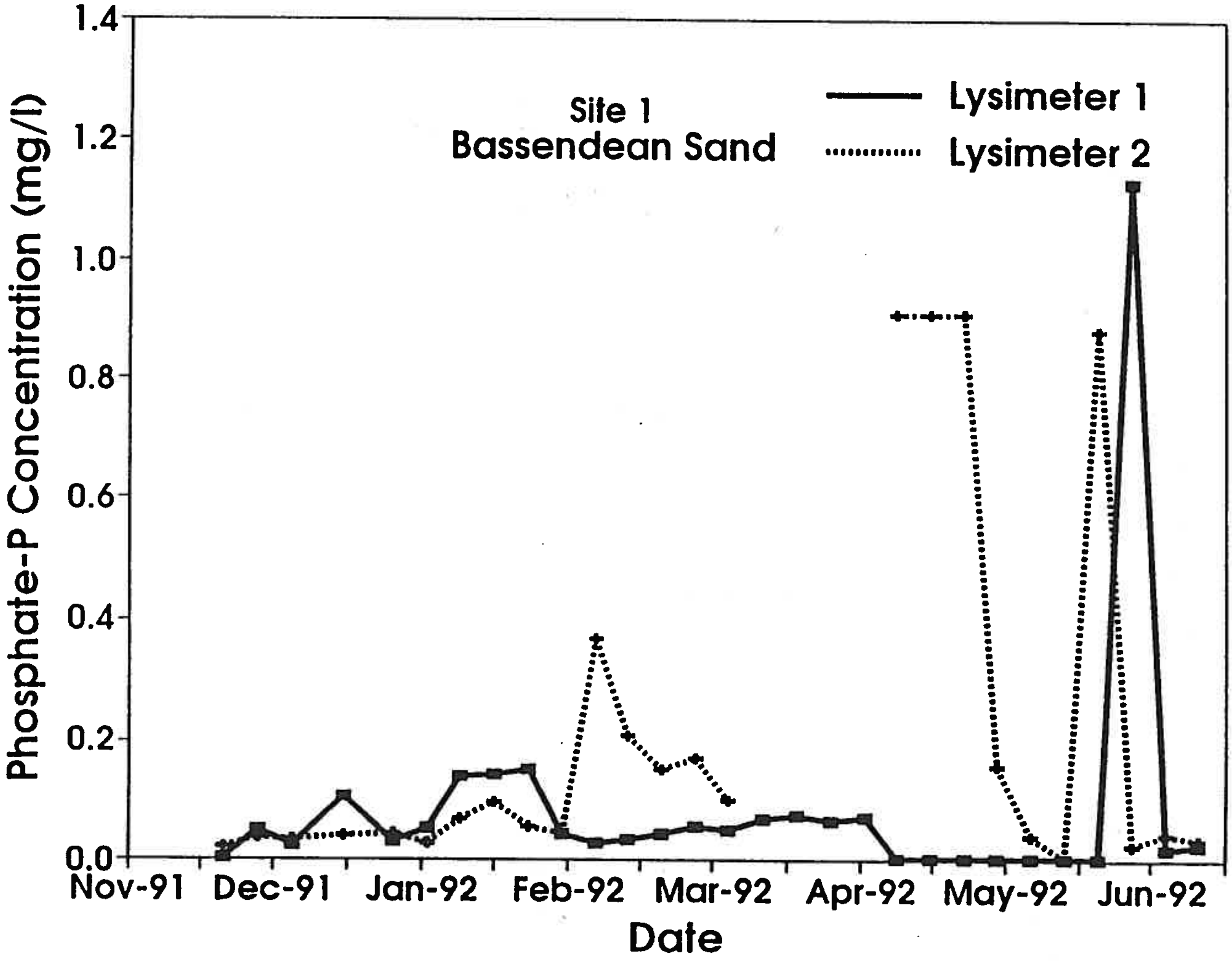

Figure 29. Phosphate-P concentration in leachate from lysimeters at urban site 1 (Noranda).

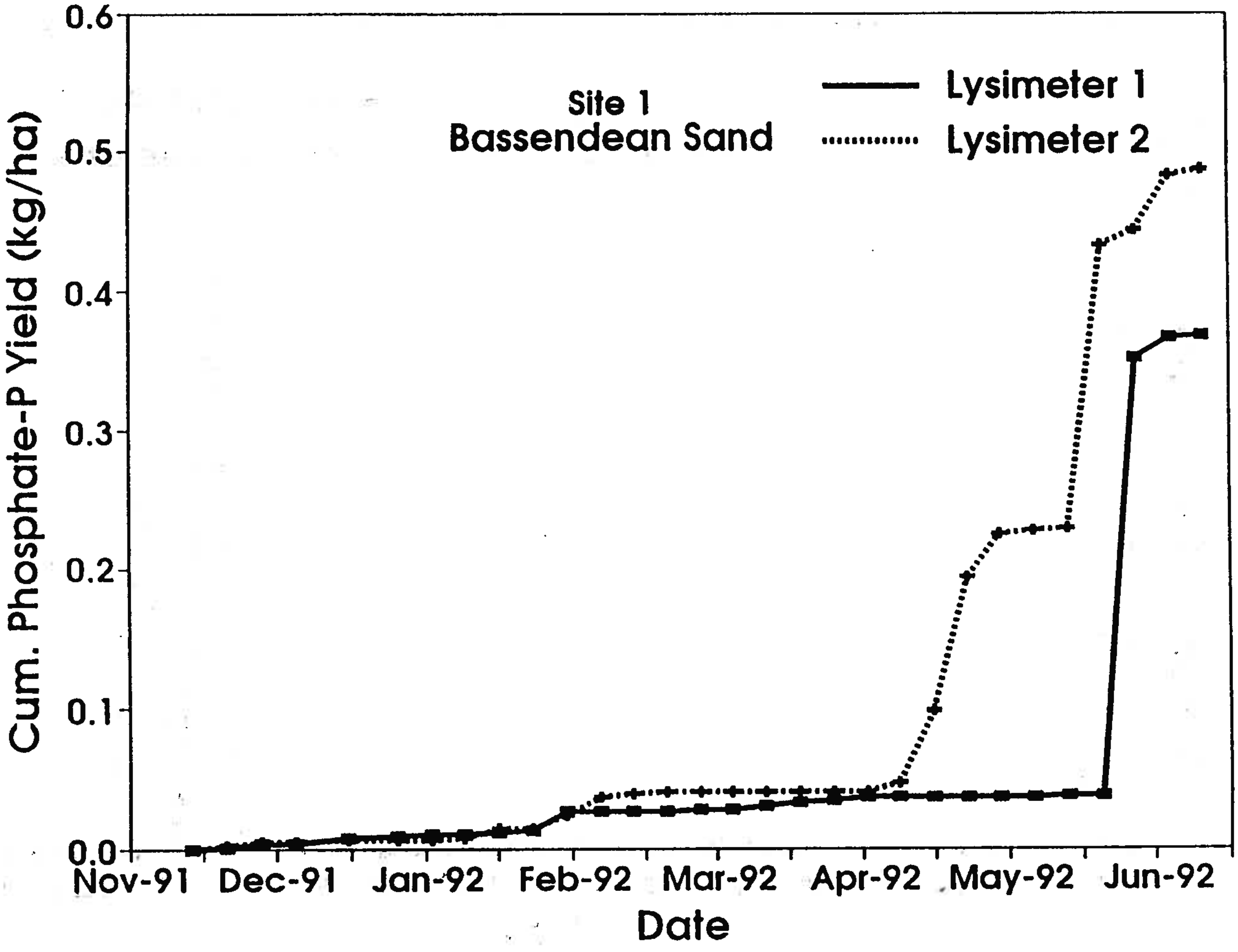

Figure 30. Phosphate-P yield in leachate from lysimeters at urban site 1 (Noranda).

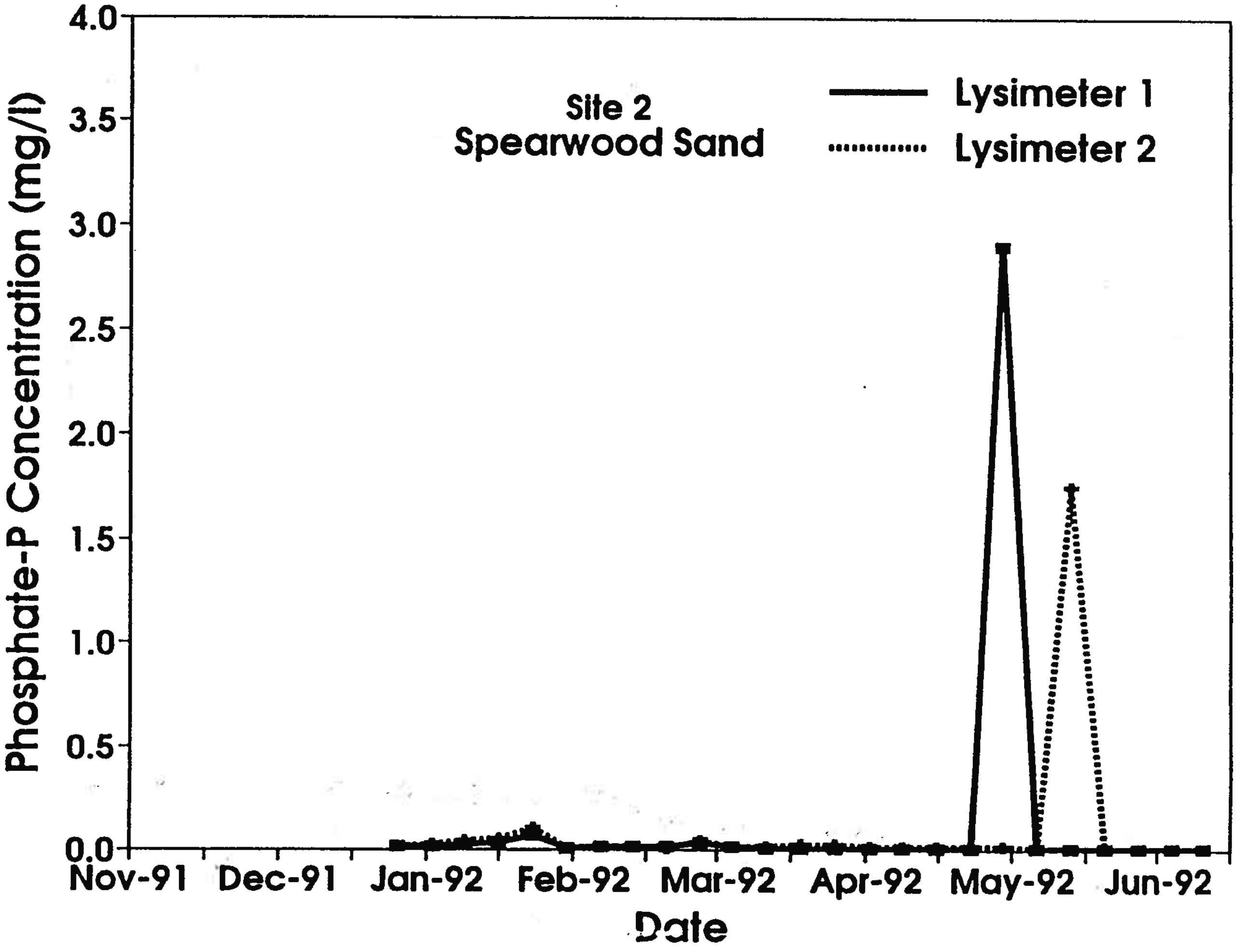

Figure 31. Phosphate-P concentration in leachate from lysimeters at urban site 2 (Tuart Hill).

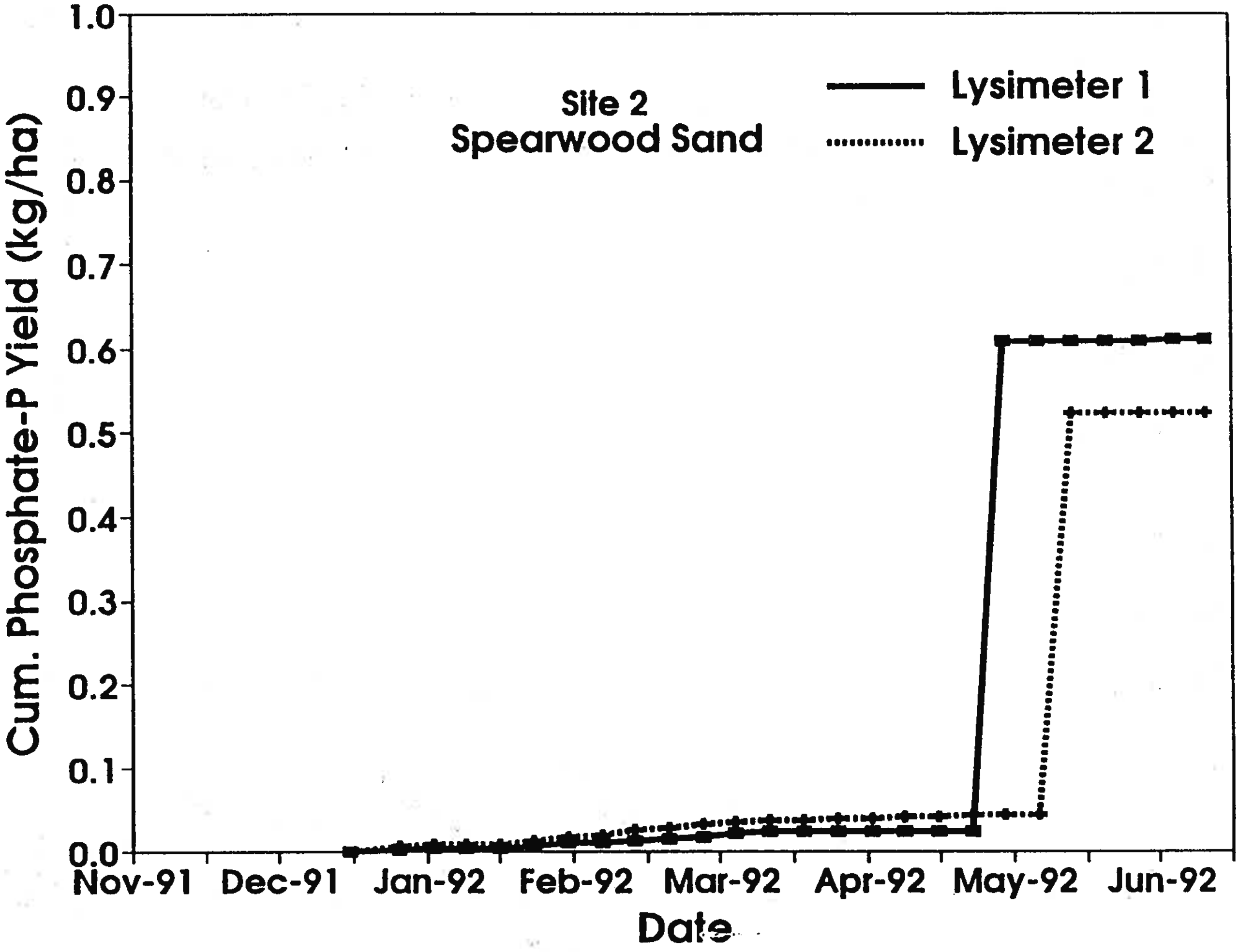

Figure 32. Phosphate-P yield in leachate from lysimeters at urban site 2 (Tuart Hill).

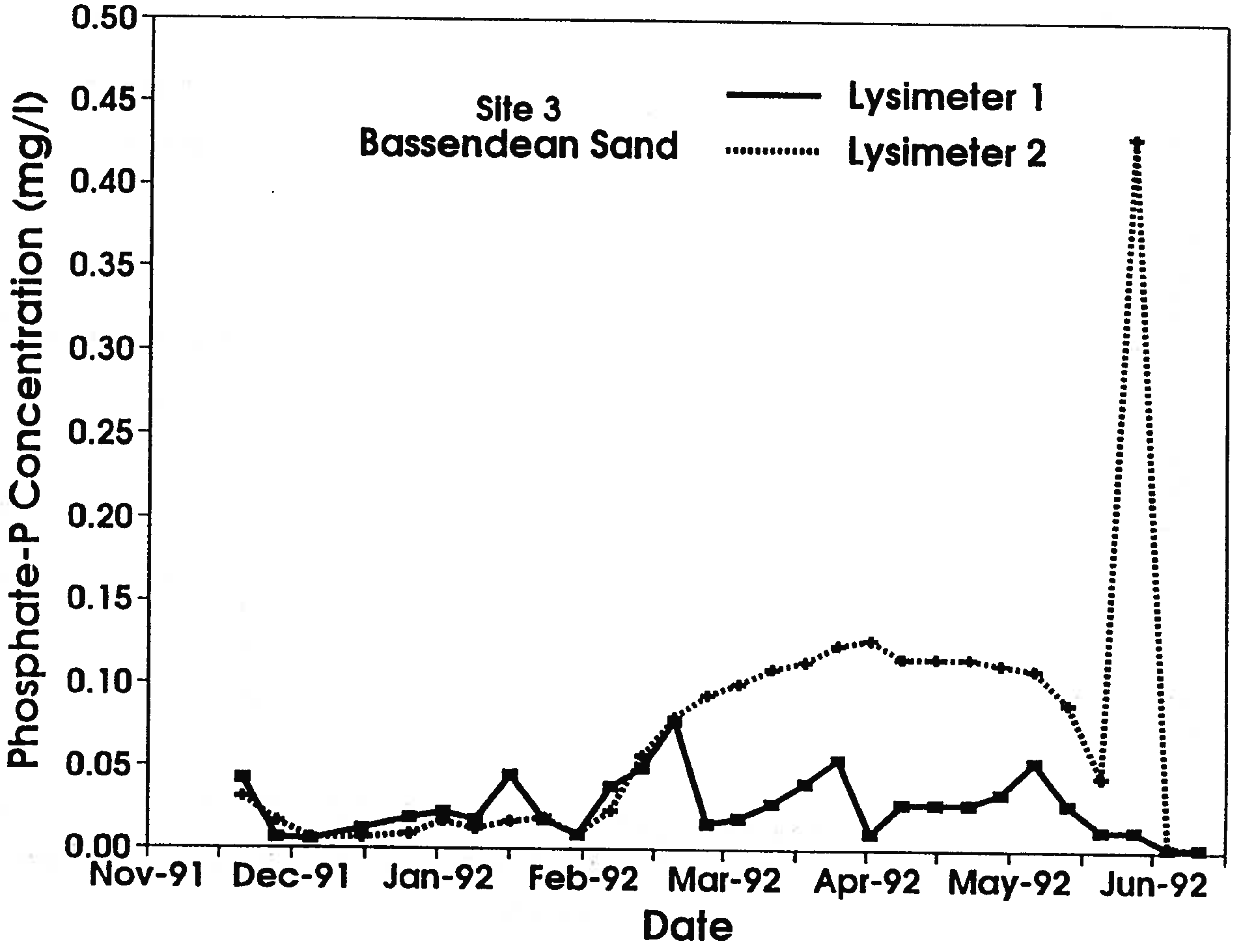

Figure 33. Phosphate-P concentration in leachate from lysimeters at urban site 3 (Mt Lawley).

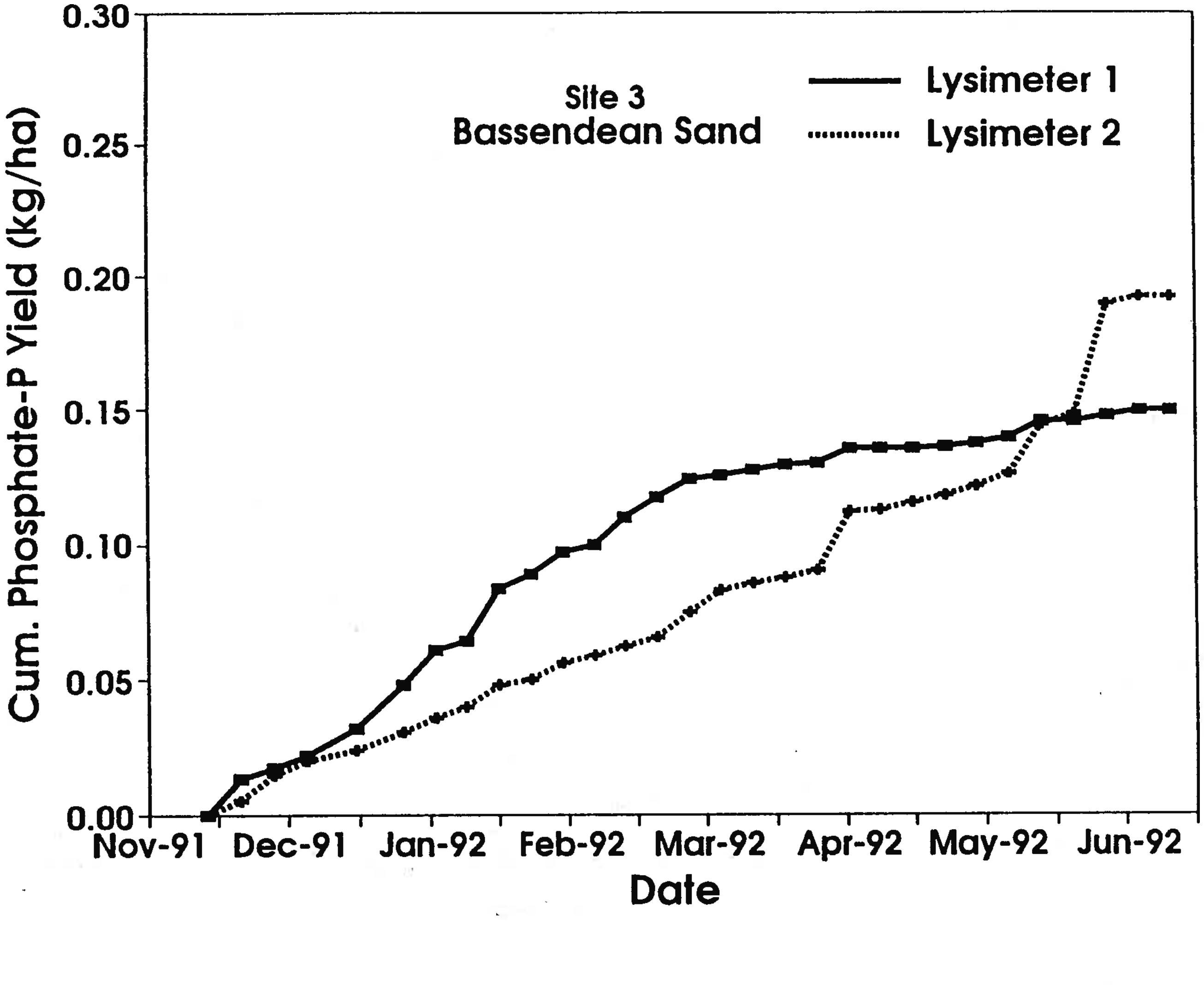

Figure 34. Phosphate-P yield in leachate from lysimeters at urban site 3 (Mt Lawley).

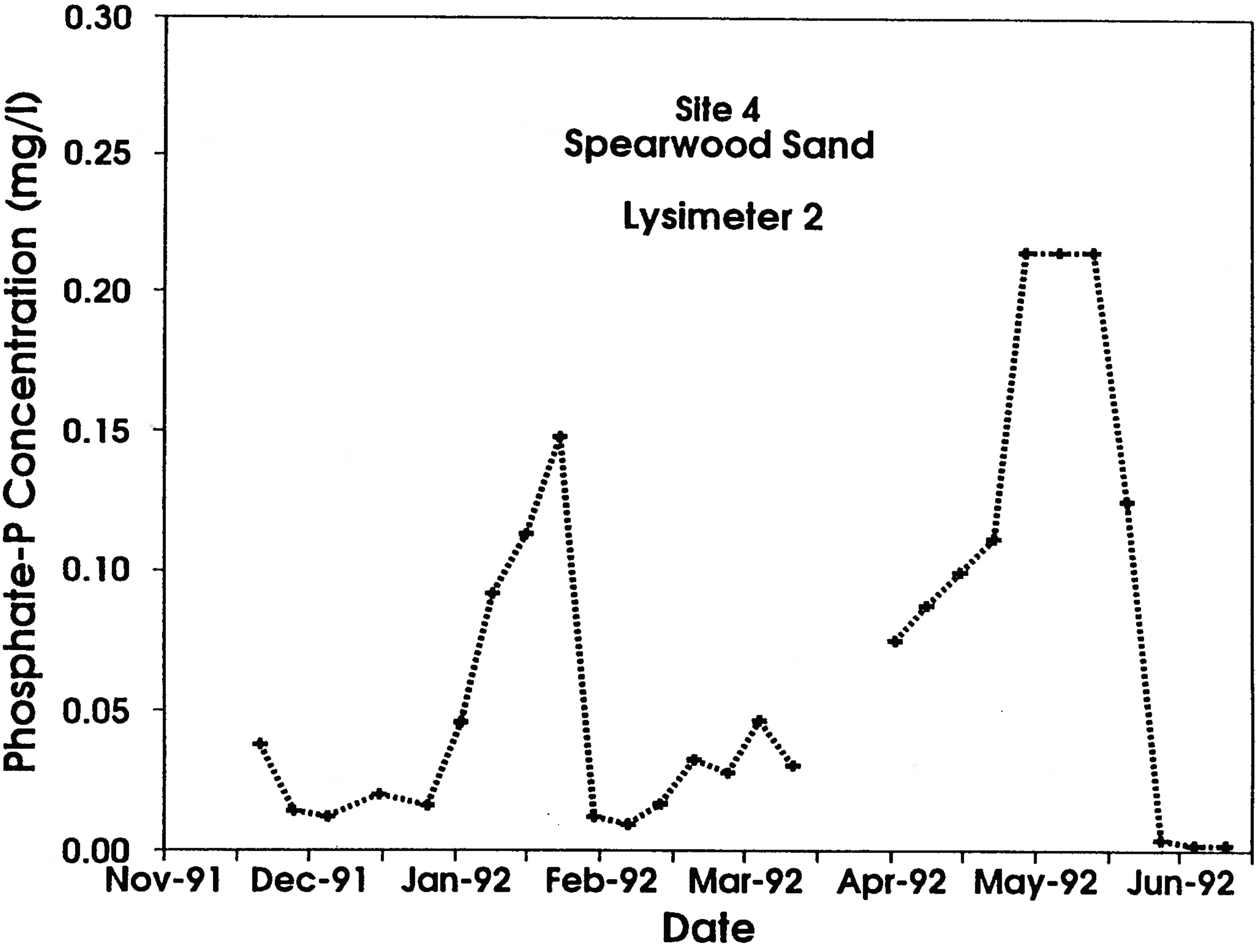

Figure 35. Phosphate-P concentration in leachate from a lysimeter at urban site 4 (Karrinyup).

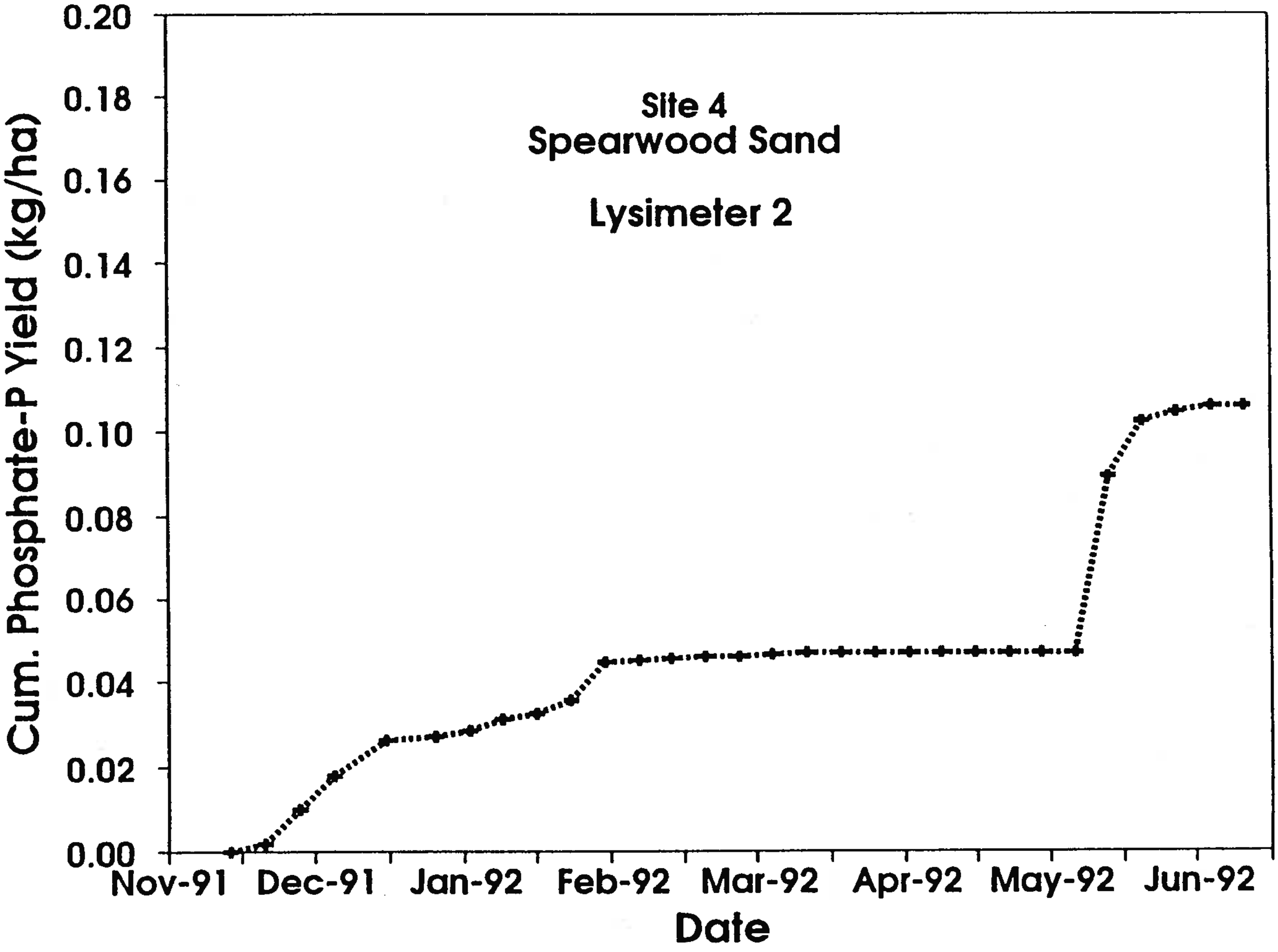

Figure 36. Phosphate-P yield in leachate from a lysimeter at urban site 4 (Karrinyup).

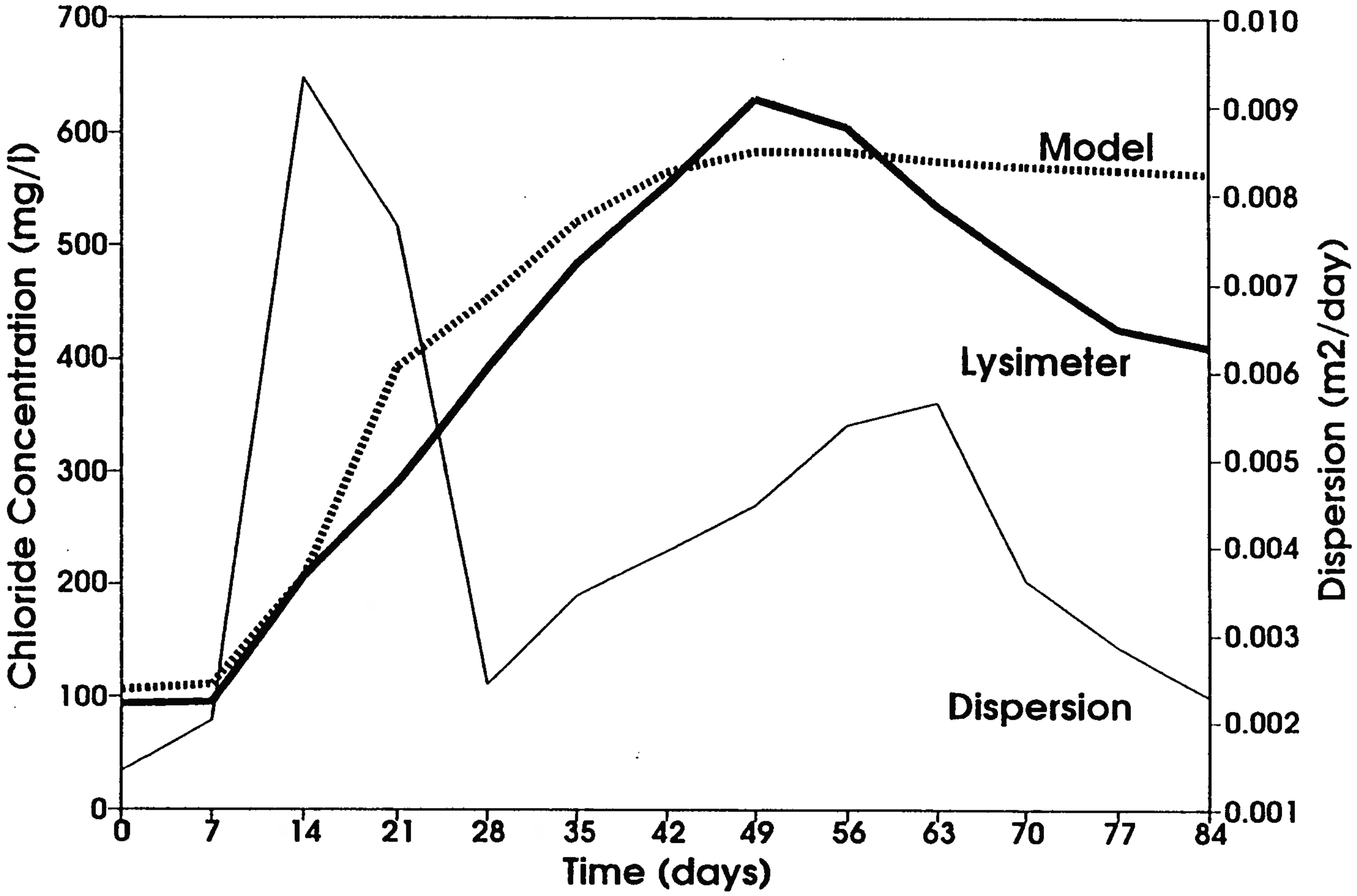

Figure 37. Comparison of chloride concentration measured by a lysimeter at 2 m depth (at a horticultural farm) and simulated by the model. The dispersion coefficient (shown) as a function of pore water velocity was obtained through the inverse parameter estimation techniques.

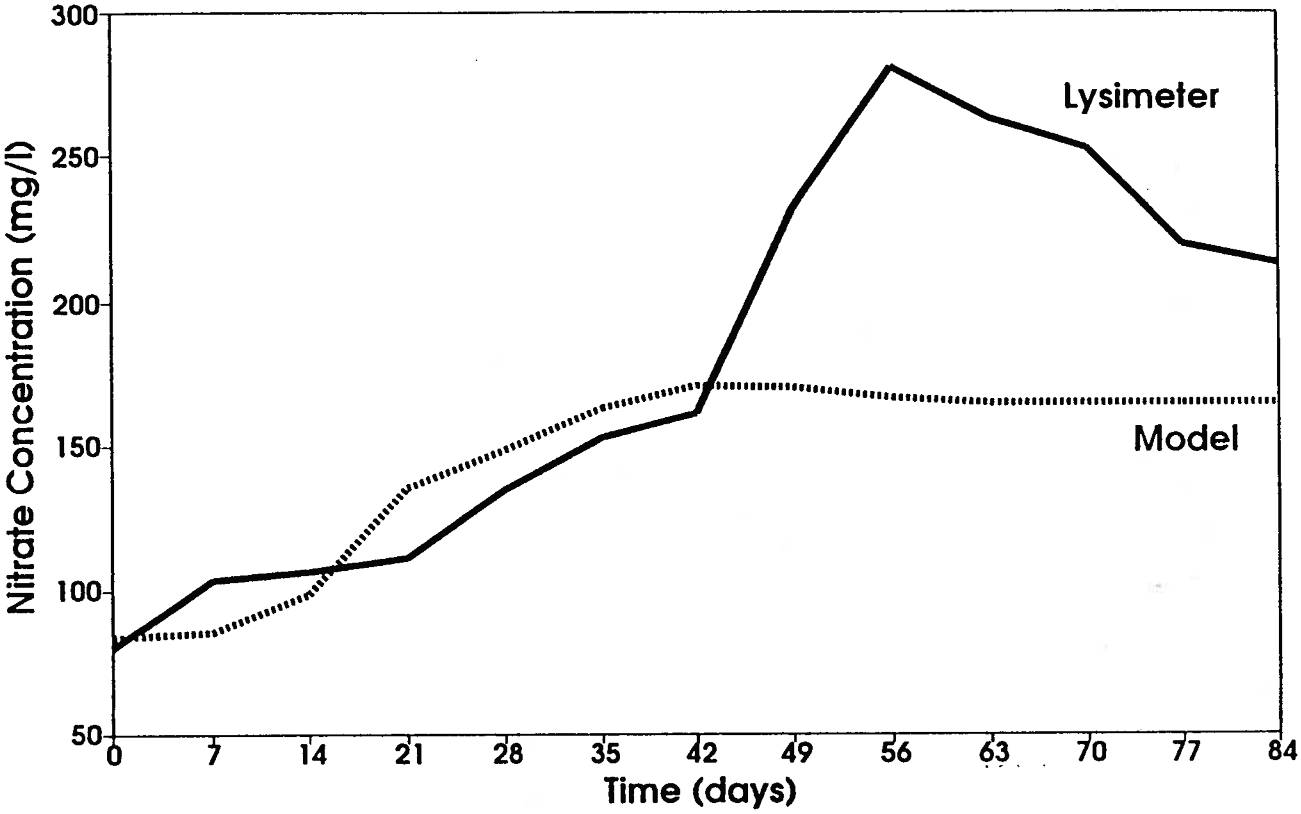

Figure 38. Comparison of measured and modelled NO_3-N concentration in the leachate at 2 m depths at a horticultural farm. The model excluded the production rate term. Dispersion/diffusion coefficients were those obtained through the inverse parameter estimated technique for a conservation tracer (Cl).

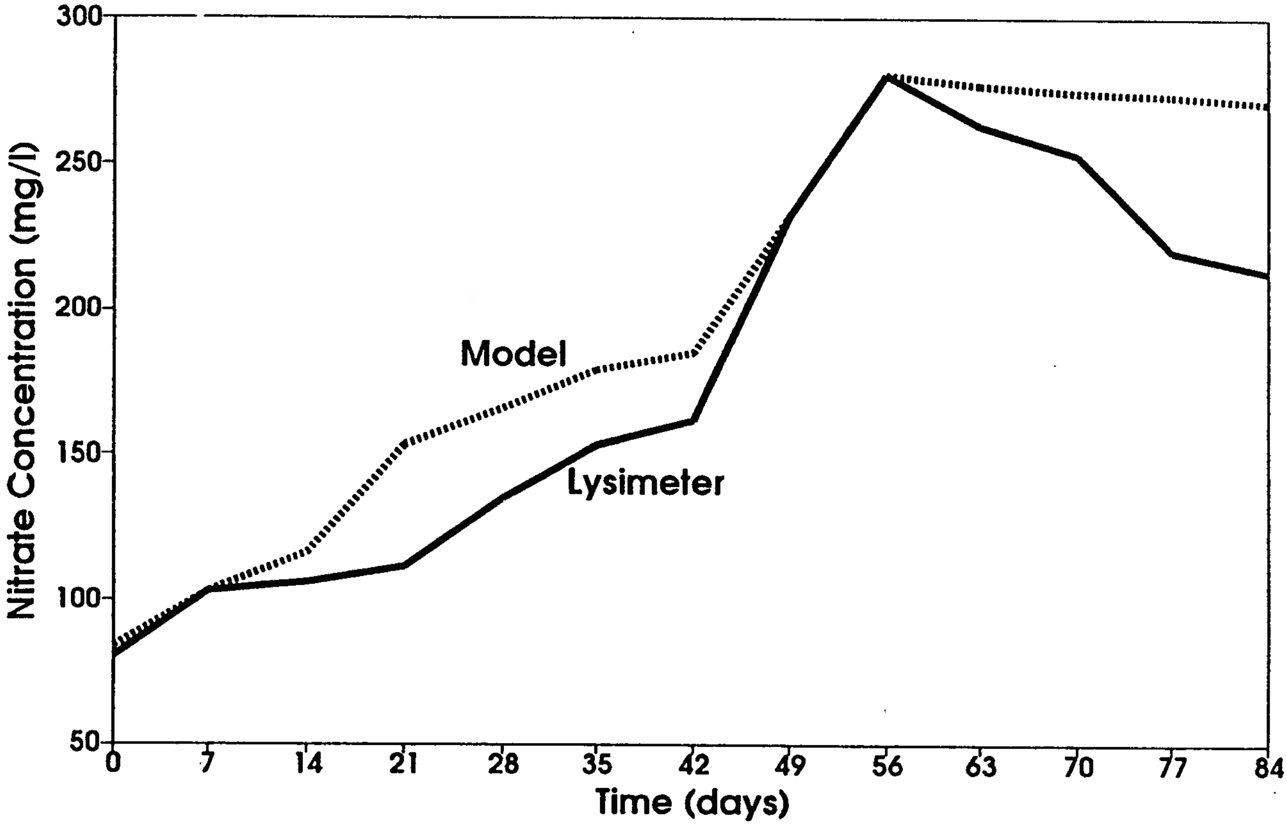

Figure 39. Comparison of measured and modelled NO_3-N concentration in the leachate at 2 m depth at a horticultural farm. The model included zero-order production rate term for NO_3. Dispersion/diffusion coefficients were those obtained through the inverse parameter estimated technique for a conservative tracer (Cl).

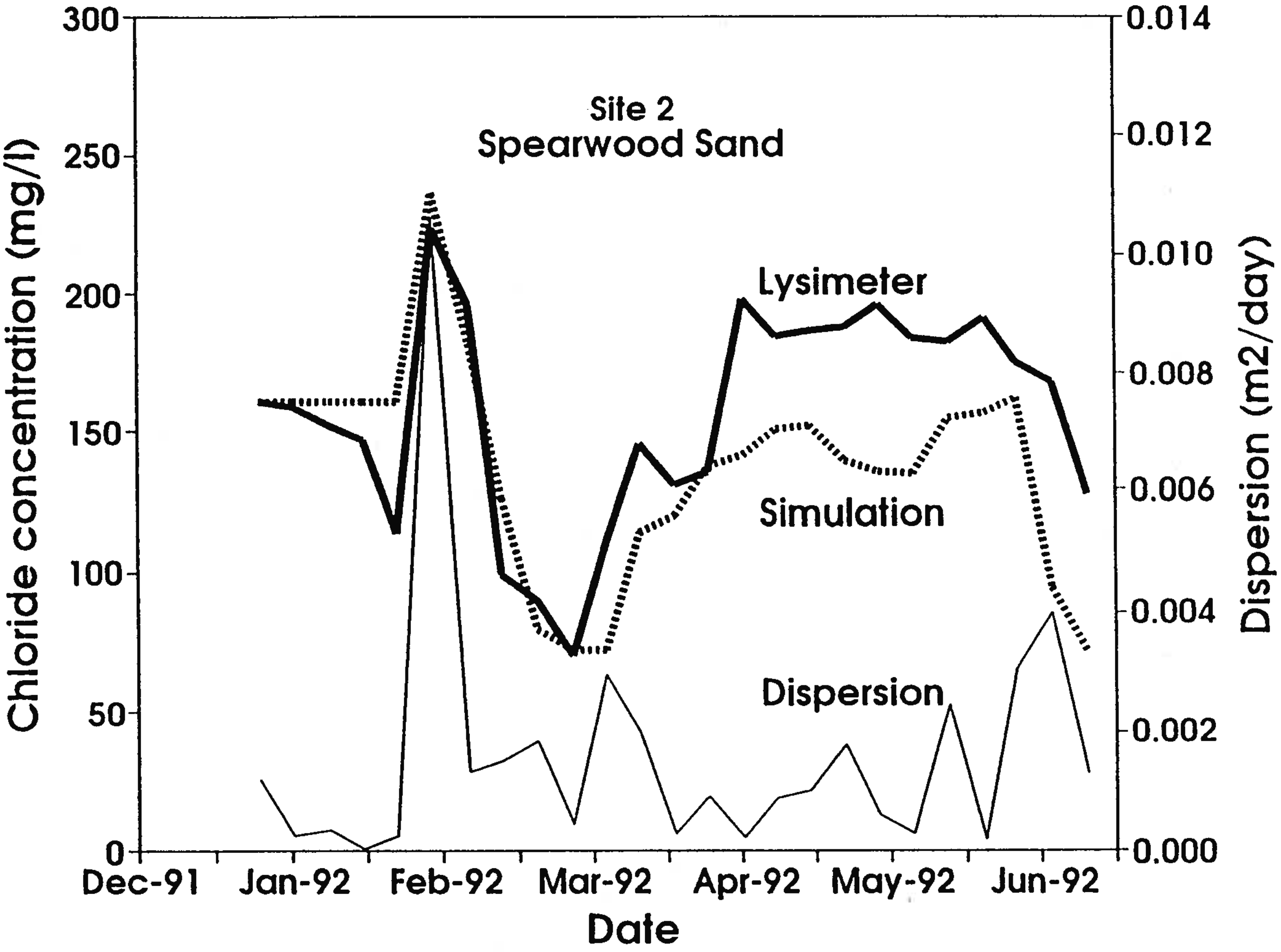

Figure 40. Comparison of measured and simulated chloride concentration in leachate at lysimeter depth at urban site 2 (Tuart Hill). The dispersion coefficient (shown) as a function of pore water velocity was obtained through the inverse parameter estimation technique.

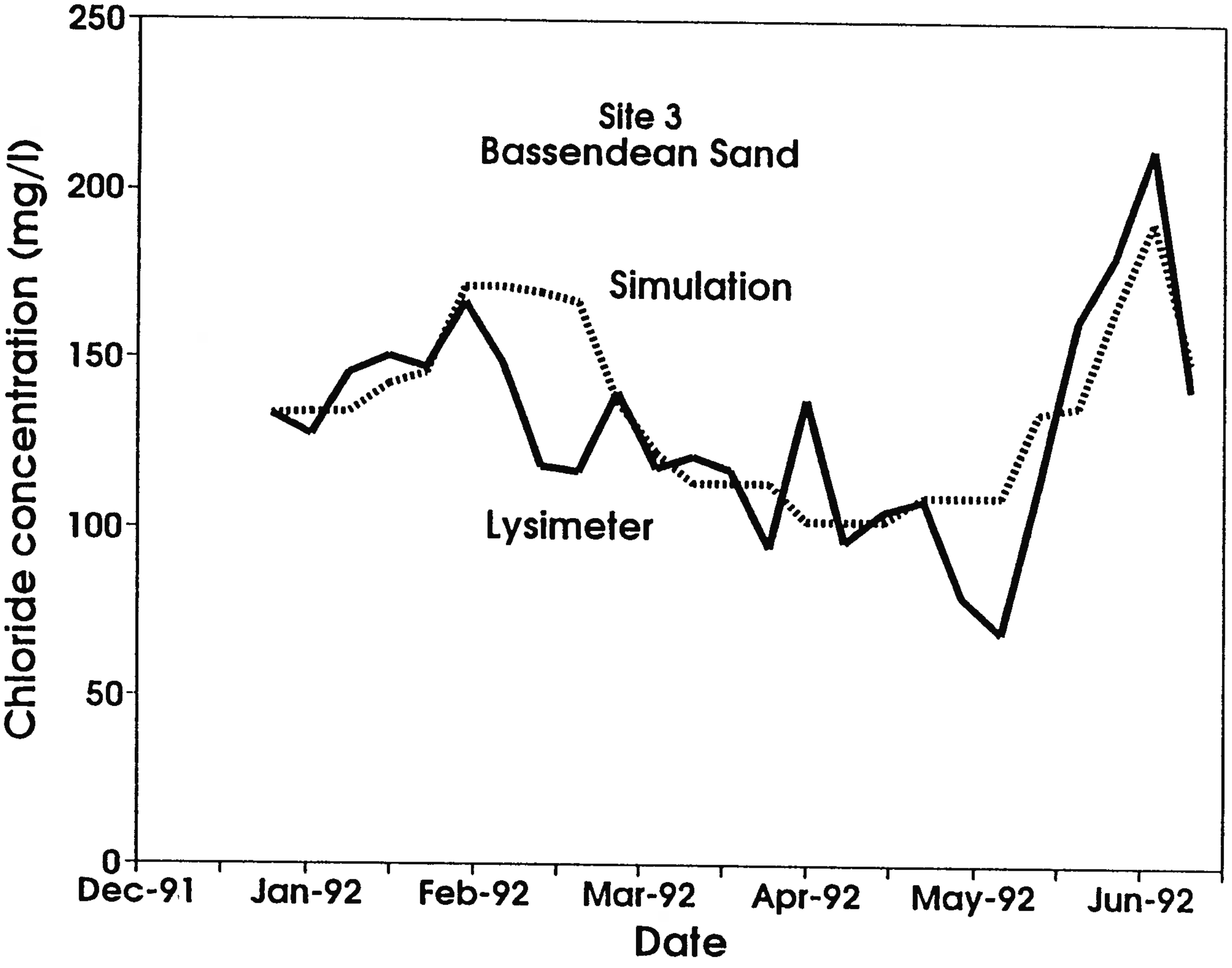

Figure 41. Model prediction of chloride concentration in leachate at urban site 3 (Mt Lawley). Observed chloride concentration in the leachate also shown for comparison. The dispersion coefficient function used in the prediction was obtained through inverse parameter estimation techniques applied to chloride data from urban site 2 (Tuart Hill).

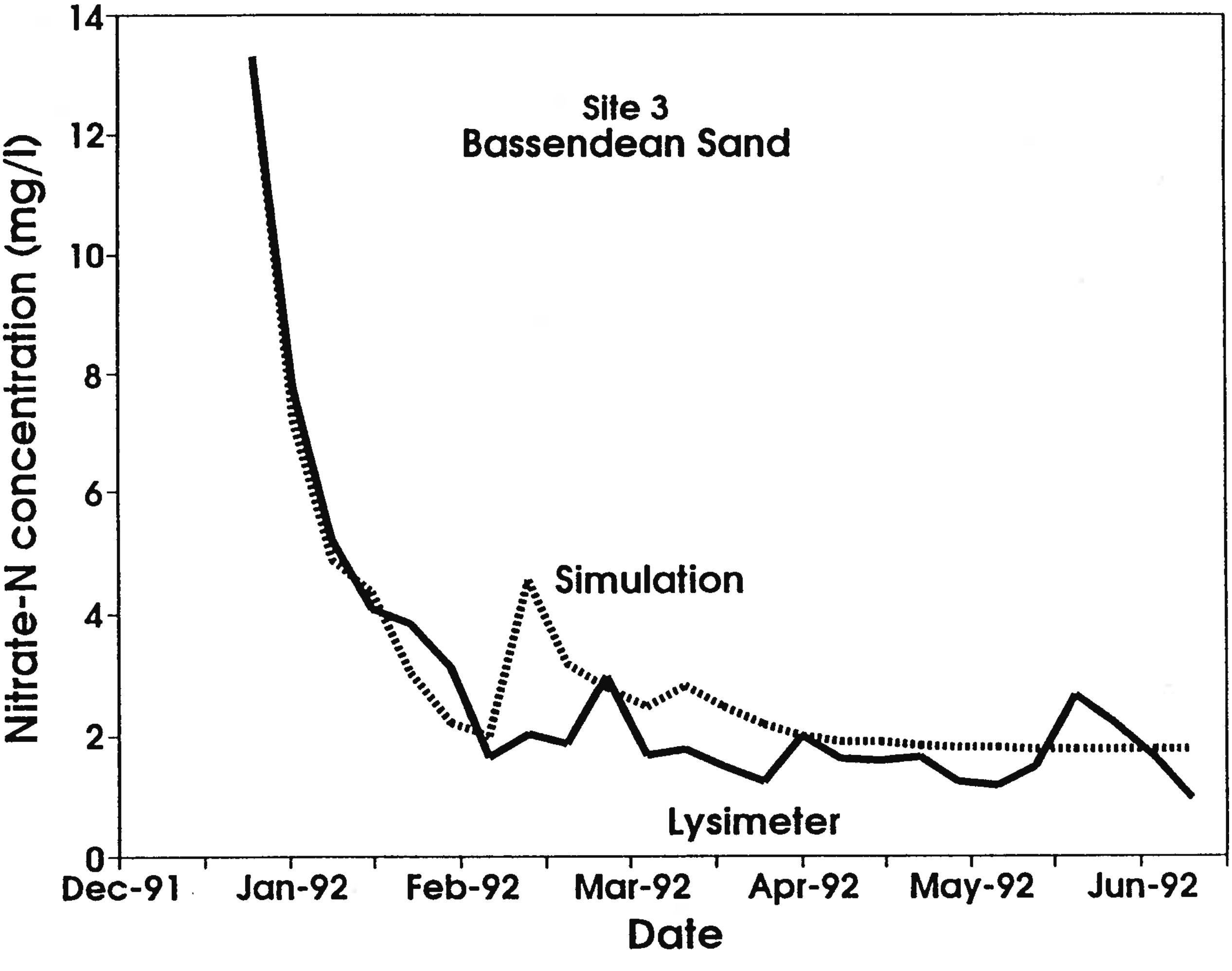

Figure 42. Model prediction of NO_3-N concentration in leachate at urban site 3 (Mt Lawley) Observed NO_3-N concentration in the leachate also shown for comparison. The dispersion coefficient function used in the prediction was obtained through inverse parameter estimation techniques applied to chloride data from urban site 2 (Tuart Hill). NO_3-N concentration profiles were dynamically adjusted by NO_3 produced through nitrification.

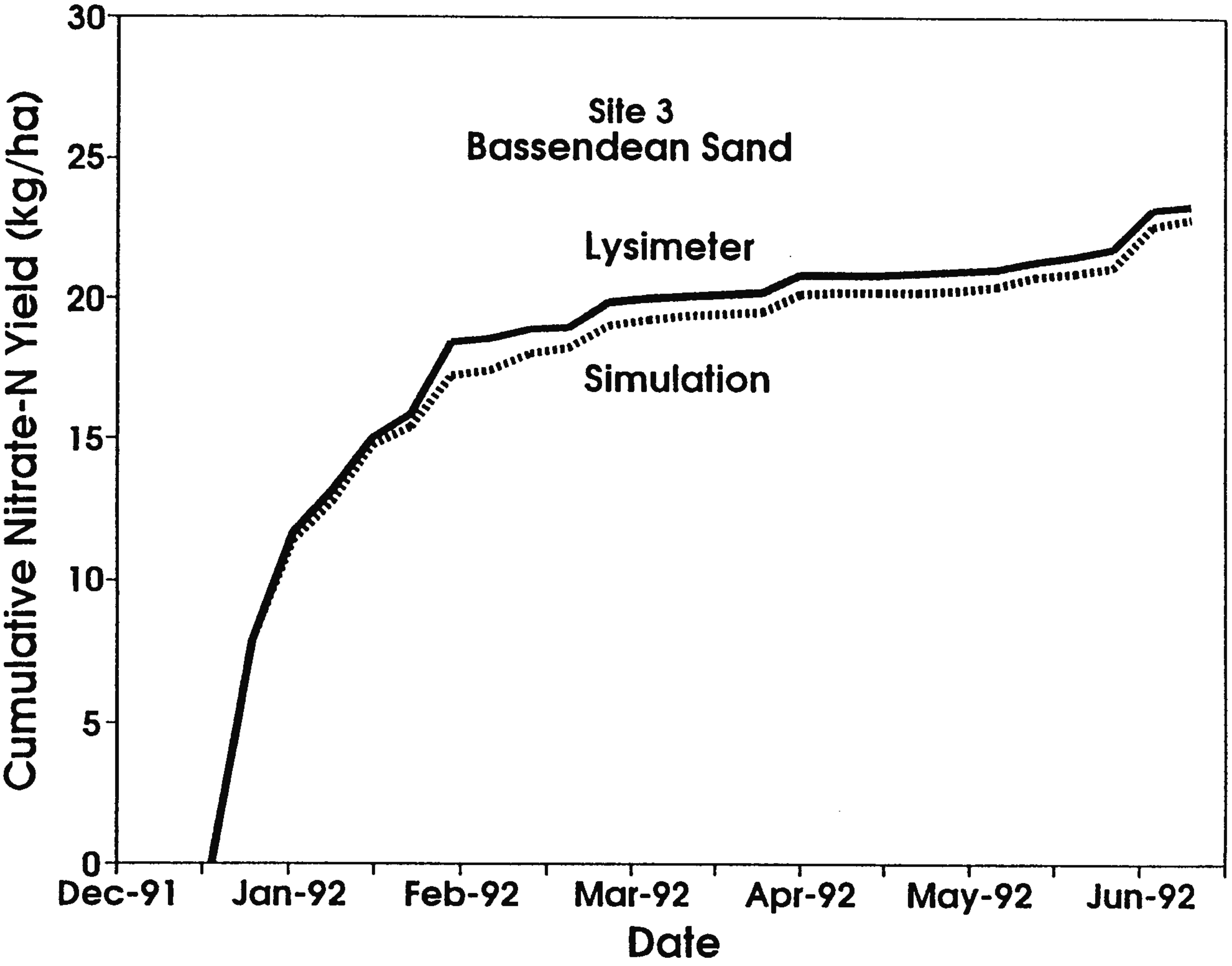

Figure 43. Comparison of cumulative NO_3-N yields calculated from lysimeter data and calculated from model predicted NO_3-N concentration at lysimeter depth for urban site 3 (Mount Lawley). The dispersion coefficient function used in the prediction was obtained through inverse parameter estimation techniques applied to chloride data from urban site 2 (Tuart Hill). NO_3-N concentration profiles were dynamically adjusted by NO_3 produced through nitrification.

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